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NEWS OF ENVIRONMENTAL RESEARCH IN CINCINNATI

1975

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16. ABSTRACT

A series of reports on the following subjects: NERC--Cincinnati electron microscope facility; poliovirus and bacterial indicators of fecal pollution in landfill leachates; examination of drinking water supplies for viruses; a copper-cadmium column for manually determining nitrate; pollution from urban land runoff; striving for zero discharge of industrial wastes; sewer transport of household refuse; a replacement for the refuse truck; the bacterial quality of bottled water; modern ways of strip mining in mountainous areas; nature and use of coal ash from utilities; a radiotelemetry system for monitoring respiration in dogs; improving the fuel value of sewage sludge; and do regulated freight rates discourage recycling?

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News of Environmental Research in Cincinnati

U.S. Environmental Protection Agency

METHODS DEVELOPMENT AND QUALITY ASSURANCE RESEARCH

January 15, 1975

NERC–CINCINNATI ELECTRON MICROSCOPE FACILITY

Robert S. Safferman*

BACKGROUND

Scientists at the National Environmental Research Center (NERC) in Cincinnati have in the past relied on other laboratories for support of ultrastructural studies. With the Center's expanded efforts in this area, efficient research management could no longer depend solely on extramural activities; in April, 1973, NERC Director Dr. Andrew W. Breidenbach authorized the establishment of the first electron microscope (EM) facility in the history of the Center. For its implementation, the Director appointed a committee of principal scientists representing the major program users of the facility. To fulfill the multi-faceted needs of the NERC–Cincinnati laboratories, the committee recommended purchase of the JEOL** model JEM 100B electron microscope (Figure 1).

Equitable use of the facility was ensured through continuation of the committee concept. Functioning in four regulatory areas, the committee enacts EM operating policies; evaluates periodically the facility's performance; assesses new or additional support equipment for purchase; and designates responsibility for maintaining the EM facility. EM utilization is unique in that each laboratory desiring use of the facility is responsible for providing its own EM expertise.

The basic JEM 100B was installed in the Robert A. Taft Laboratory, where it will be temporarily housed until the new NERC complex adjacent to the University of Cincinnati campus is completed. Major modifications have been made on the microscope since its initial installation as a transmission mode instrument. An added high-resolution scanning attachment (Figure 2) provides, in the combined EM system, the capability of specimen examination by scanning, scanning transmission, and conventional transmission modes. A side-entry goniometer (Figure 3) has replaced the original top-entry stage to permit relatively large-angle specimen tilting needed for proper orientation to the detector of the Ortec energy dispersive X-ray spectrometer. This spectrometer serves to determine elemental composition directly from EM observed specimens.

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**Japan Electron Optics, Ltd. (Mention of trade names or commercial products does not constitute endorsement or recommendation for use.)

RESEARCH APPLICATION

To many NERC–Cincinnati programs, the new EM facility has already opened new realms of research. Although it is too early to appreciate its full effect in realizing EPA objectives, the EM facility has important applications for a variety of major environmental problems in the areas of advanced waste treatment, methods development, water and air toxicology, and water supply. Obviously, high-resolution EM studies are indispensable for structural determinations of virus-infected cells, virions, isolated viral nucleic acids and proteins, and small parasitic bacteria such as *Bdellovibrio*. High-resolution microscopy is extremely important in inhalation toxicology programs concerned with ultrastructural evaluation of tissue morphology after animal exposure to smoke and dust particulates. Of prominence recently has been the use of EM in water standards development, where it has been used to identify the presence of high numbers of asbestos fibers in drinking water supplies. Elemental analysis of colloidal material remaining after wastewater

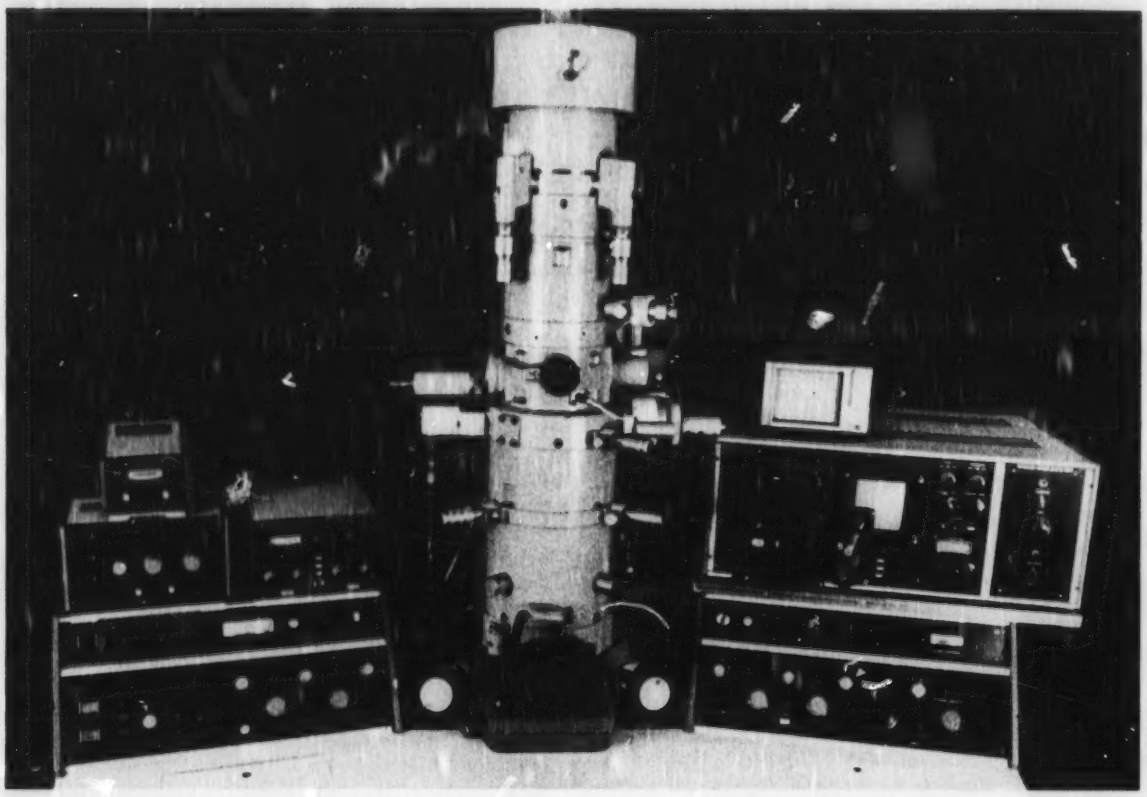


Figure 1. JEM 100B electron microscope operable in scanning, scanning transmission, and conventional transmission modes. The EM is housed in a 12'x18' room previously determined to be free from EM-interfering mechanical vibrations and magnetic fields. Also utilized are a connecting darkroom and specimen preparation laboratory containing such EM accessory equipment as light microscopes, vacuum coater, and ultramicrotome.

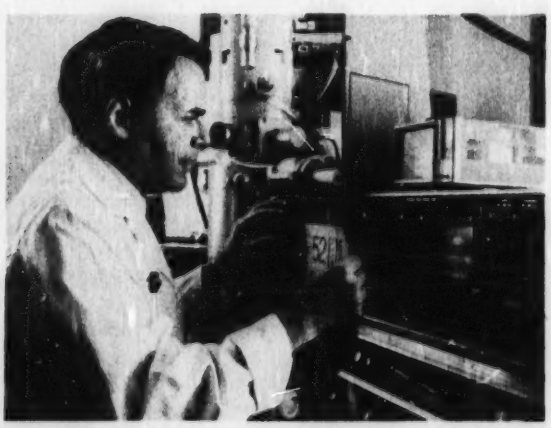


Figure 2. Recording EM image from high-resolution scanning device.

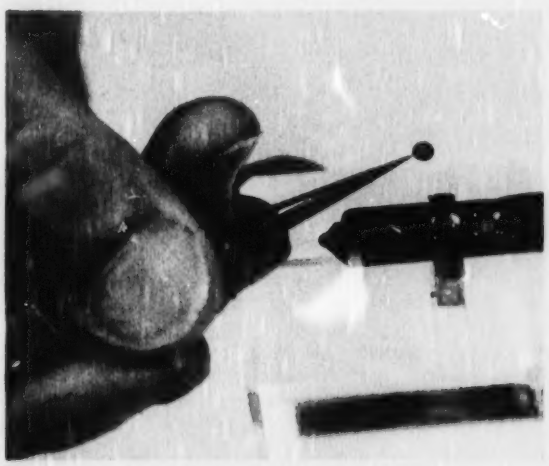
treatment and ultrastructural study of polymer attachment in flocculation and sludge conditioning are other less prominent, but potentially important, EM applications in water management.

In support of quality assurance needs, EM studies have been programmed to determine the degree of viral particle aggregation in suspension so that the observed number and state of these aggregates can be interpreted with regard to disinfection kinetics and virus-cell interactions. Further viral studies are intended to determine size and kind of particulates (waterborne sediments, silts, fibers, and sludges) involved in viral binding and to elucidate the mode of virus attachment as well as the effect of environmental factors on this binding.

To ascertain the toxic effects of air and/or water pollutants from mobile and stationary sources,

EM studies have also been programmed to delineate ultrastructural changes in animal tissues. Such studies involve multimedia exposure to a combination of organic agents, which include polynuclear aromatic hydrocarbons, halo ethers, and chlorinated hydrocarbons; and to single pollutants such as trace metals resulting from industrial discharges (mercury and cadmium), water treatment practices (copper), the use of fuel additives (lead and manganese), and use of air pollution control devices (degradation products from platinum, palladium, and aluminum).

In the first months that the EM has been in service, about half the time has been allotted to staff training. The remaining time has been applied to the ultrastructural study of such biological material as viruses (Figure 4) and parasitic bacteria (Figure 5) and to the analysis of asbestos fibers (Figures 6 and 7). With the acquisition of the energy dispersive system, the EM is being used in its first function as a total analytic tool to evaluate the distribution, size, kind, and mineral composition of asbestos fibers found in drinking water throughout the United States.



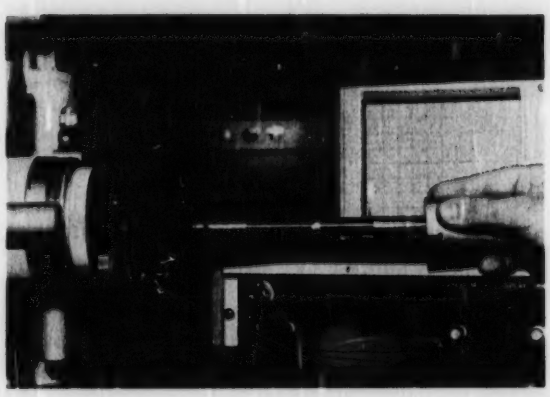


Figure 3. Positioning of specimen: (Top) insertion of grid in rotating specimen holder. (Bottom) insertion of holder through airlock into side-entry goniometer.

Figure 4. Electron micrograph of AS-1 virus that infects the unicellular blue-green algae *Anacystis nidulans* and *Synechococcus cedrorum*. Bar represents 1000 A. (R. Safferman)

Figure 5. Electron micrograph of a parasitic bacterium, *Bdellovibrio bacteriovorus*, penetrating its host cell. Bar represents 5000 A. (A. Venosa)

SUMMARY

The instrument, now totally operable for chemical identification and structural determination, fully supports the research objectives proposed for the EM facility. In its future applications, the EM should be as much a necessity to analytical aspects of environmental research as the gas chromatograph and mass spectrometer.

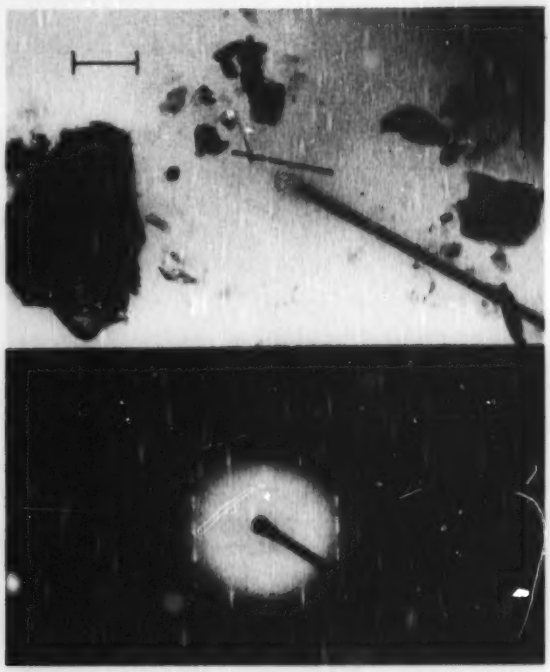


Figure 6. Chrysotile asbestos in raw water sample: (top) morphological observation of particle, (bottom) selected area diffraction pattern. Bar represents 5000 A. (R. Lishka)

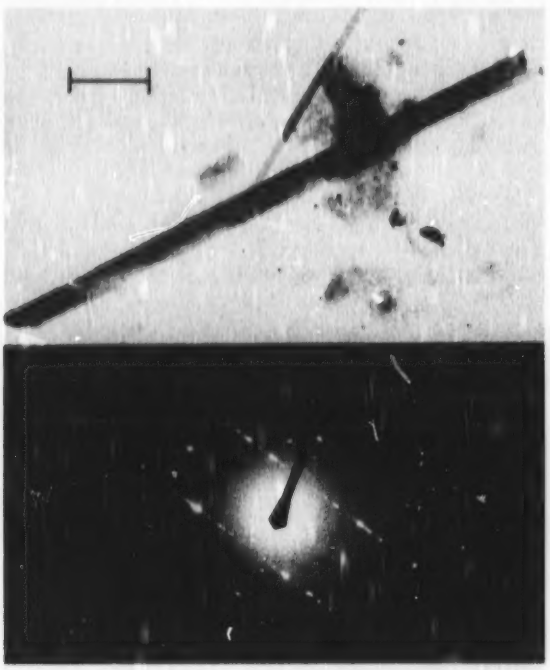


Figure 7. Amphibole asbestos in raw water sample: (top) morphological observation of particle, (bottom) selected area diffraction pattern. Bar represents 5000 A. (R. Lishka)

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SOLID AND HAZARDOUS WASTE RESEARCH

January 31, 1975

POLIOVIRUS AND BACTERIAL INDICATORS OF FECAL POLLUTION IN LANDFILL LEACHATES

R. A. Glotzbecker and A. L. Novello*

BACKGROUND

One of the basic objectives of solid waste disposal is the protection of public health and environment. Presently, over 90 percent of solid waste in the United States is disposed of on land. One public health concern is the presence of untreated human and animal fecal matter in solid waste disposal sites. Excreta and products of animals have long been a part of municipal solid wastes;[3] the use of disposable diapers is one way human excreta become present in solid waste. Viral and bacterial pathogens associated with fecal matter in solid waste represent health hazards because of their potential for leaching from the landfill to ground or surface waters.

STATE OF THE ART

The degree of the health hazard from enteric infectious agents in sanitary landfills depends on three basic conditions:[5] the amount and nature of the pathogens in the disposed solid waste, the survival of the pathogens in the landfill environment, and the movement of the pathogens from the landfill into the environment (e.g., groundwater).

Studies have shown that raw municipal solid waste can contain fecal coliforms, fecal streptococci, and pathogens such as salmonella and enteroviruses.[4,7,8] Little is known about the survival of enteric pathogens in the landfill environment, but fecal coliforms, fecal streptococci, and enteroviruses have been found in leachates.

During the initial 2-month leaching period of a 10-month study by Blannon and Peterson,[1] an experimental field-scale landfill leachate contained high densities of fecal coliforms (averaging 1.5 million and 48 million organisms per 100 ml, respectively). After 2 months, the fecal coliform density dropped to low values that continued through the remainder of the study. Fecal streptococci, however, were leached in high, although quite variable, numbers throughout the study.

Engelbrecht[5] used a lysimeter containing shredded municipal solid waste to simulate a sanitary landfill. Poliovirus, reovirus, and Rous sarcoma were added as the lysimeter was filled with solid waste. Fecal coliforms and fecal streptococci were detected in the leachate and showed some persistence before disappearing rapidly after 76 to 96 days of lysimeter operation. No viruses were detected in leachate samples using direct plaquing techniques.

The survival of poliovirus, fecal streptococci, and *Salmonella typhimurium* was also investigated by Engelbrecht[5] in leachates at pH 5.3 and 7.0 incubated at 22 C and 55 C. Bacteria and viruses were more rapidly inactivated at 55 C than at 22 C; such high temperatures are not uncommon during initial solid waste decomposition in a sanitary landfill. Ambient temperatures (10 C to 20 C) are observed at the periphery of the landfill and after the anaerobic fermentation processes predominate. The rate of inactivation was greater at pH 5.3 than at pH 7.0; leachate from a landfill is typically between pH 5.0 and 5.5 because of the vigorous production of organic acids. These results indicate that acidic leachates and the initial high temperatures of a landfill can inhibit bacterial and viral survival.

Cooper *et al.*[4] conducted a study involving simulated sanitary landfills and simulated open dumps. Bacteriological analysis indicated the presence of large numbers of indicator bacteria of fecal pollution in the leachates. The coliform density in sanitary

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landfill lysimeter leachate declined relatively rapidly with time; coliform density declined much less rapidly in open dump leachates. Virus recovery from the leachates was generally low and irregular over a period of 20 weeks. The virus recovery technique was a modification of a method developed by Sobsey *et al.*[11] The use of sodium ethylenediaminetetraacetate (EDTA) and the dilution of preclarified leachate with water facilitated successful virus concentration. Cooper *et al.* noted that without the addition of EDTA to the cell culture inoculum, the results would erroneously indicate that the leachates were highly viricidal. The results of the study indicated that gradual saturation of landfills may cause greater inactivation of viruses than would rapid saturation. This detrimental effect did not seem to apply to the bacterial parameters.

CURRENT RESEARCH

The Solid and Hazardous Waste Research Laboratory recently completed an investigation to determine the occurrence of bacterial indicators of fecal pollution in leachates, the survival of these bacteria and poliovirus in leachates, and the removal of bacteria and viruses in leachates by soils.

Occurrence of Indicator Bacteria of Fecal Pollution in Leachates

The results of a comparative study indicated that the most probable number (MPN)[12] method was superior to the membrane filter (MF)[12] method for determining fecal coliforms and fecal streptococci in a municipal landfill leachate. Fecal coliform and fecal streptococci densities in municipal landfill leachates determined by the MPN method were 10 to 1,000 times higher than those determined by the MF method. Similar studies by Smith and Madison[10] indicate the same conclusion. Others[2] have recognized the limitations of the MF method in other types of waters. The high, heavy metal cation concentration typically found in leachates, is believed to be one of the factors that limited the use of the MF method. Based on these results, the MPN method was used throughout this study.

Leachates from an active municipal landfill that received some 1,800 metric tons (2,000 short tons) (wet weight) of waste per day were bacteriologically examined. The densities of fecal coliforms and fecal streptococci from this municipal landfill leachate are shown in Table 1. Leachates from an experimental landfill completed in 1971 were also examined bacteriologically. This experimental landfill study (the Boone County Field Site) was a continuation of the work done by Blannon and Peterson. The Boone County leachate initially contained high densities of fecal coliforms and fecal streptococci; unlike fecal streptococci, the fecal coliform density in leachate drastically dropped after 2 months of leaching. Fecal streptococci densities remained high, but quite variable, throughout the 35-month study. Both of these studies indicated that landfill leachate can be a potential source of enteric pathogens.

TABLE 1. FECAL COLIFORMS AND FECAL STREPTOCOCCI IN MUNICIPAL LANDFILL LEACHATE

	Organisms/100 ml (MPN)	
Sampling Date	Fecal coliforms	Fecal streptococci
1/16/74	4.6×10^2	1.5×10^3
1/30/74	7.7×10^1	1.8×10^2
2/13/74	4.6×10^2	1.4×10^1
2/28/74	1.7×10^2	2.2×10^2
3/02/74	1.1×10^1	1.4×10^2
4/02/74	9.4×10^2	2.2×10^3
5/06/74	2.3×10^1	1.1×10^2

Survival Studies

The survival of *Escherichia coli* ATCC 11229, *Streptococcus faecalis* SEC (Sanitary Engineering Center), and poliovirus 1 Mahoney (LP) RKP-42 was determined in leachates. Virus determinations were made by the plaque method, with the addition of EDTA to the cell culture inoculum, a technique developed by Sobsey *et al.*[11] Our results (similar to those of others[4,11]) indicated that EDTA improved the recovery of viruses in leachates.

Results of bacterial survival studies (Table 2) indicated that *E. coli* and *S. faecalis* may survive in leachate from hours to months depending on the leachate source. It appears that the rate of bacterial reduction in leachate depends upon temperature, leachate quality, and other factors not evaluated.

TABLE 2. DAYS FOR 99.9 PERCENT REDUCTION OF MICROORGANISMS IN LEACHATES AT 10 AND 20 C

Microorganism	Experimental landfill leachate	Municipal landfill leachate	
	10 C	10 C	20 C
E. coli	0.12	56	21
S. faecalis	0.21	>100	35

The results of the poliovirus survival studies (Table 3) indicated that virus inactivation was directly related to temperature. There was less variability of poliovirus survival than of bacterial survival in the two different leachates. The results also indicated that the use of fecal coliforms and fecal streptococci as indicators of animal viral pollution in leachates is of questionable validity, since these bacteria were very sensitive to the experimental landfill leachate.

TABLE 3. PERCENT POLIOVIRUS SURVIVAL IN LEACHATES AT 10 AND 20 C

Days	Experimental landfill leachate		Municipal landfill leachate	
	10 C	20 C	10 C	20 C
7	100	64	66	30
21	53	18	26	4.1
35	67	9.1	29	1.2
55	30	1.5	14	0.06
90	36	0.43	8.9	>0.01

Results of bacterial survival investigations indicated that the fecal coliform and fecal streptococci levels in leachates may drop drastically during transport from the sampling point to the laboratory. Leachate from experimental landfills was collected in duplicate; one sample bottle contained sufficient EDTA to give a 0.01 M concentration, and one contained none. The chelation of metal cations in leachate by EDTA protected the indicator bacteria of fecal pollution against inactivation during transport to the laboratory. Shipe and Fields[9] and Coles[3] reported similar results on the protective effect of EDTA on coliform bacteria in waters with high metal cation concentrations.

Soil Column Studies

Soils from two sanitary landfill sites were packed in 30- by 10-cm-ID Plexiglass columns to a height of 10 cm. In one column, a yellowish-brown, silty clay soil consisting of 6 percent sand, 58 percent silt, and 36 percent clay by dry weight was packed to a dry density of 1,570 kg/m^3 (97.7 pcf). In another column, a brown, gravelly, silty sand soil consisting of 20 percent gravel, 54 percent sand, 18 percent silt, and 8 percent clay by dry weight was packed to a dry density of 1,900 kg/m^3 (124 pcf). The soil columns were saturated from the bottom with sterile water to displace air.

To simulate conditions below a sanitary landfill, the columns were maintained at 10 C under an atmosphere of nitrogen. A constant head of 10 cm of leachate seeded with 10^8 *E. coli* per 100 ml was maintained above the clay soil column, and the same head of leachate seeded with 10^8 *E. coli* and 10^5 poliovirus per 100 ml was maintained above the sandy soil column. The seeded leachates and percolates were examined for fecal coliforms and viruses.

As leachate percolated through the clay soil column, *E. coli* was effectively retained in the soil matrix. Less than three fecal coliforms per 100 ml were detected in the percolate. Over a period of 119 days, 640 ml of percolated leachate were collected; this leachate represented 1.8 times the calculated pore volume of the soil and 27 times the calculated specific yield of soil.[6]

More than 99 percent of the *E. coli* and 80 percent of the poliovirus in leachate were removed during percolation through a sandy soil column. Figure 1 depicts the percent survival of the *E. coli* and poliovirus in the applied leachate and the percent recovered in the percolate. The increased breakthrough of the bacteria from days 14 to 20 may have been caused by channeling in the soil. During the study, 4.9 liters of percolated leachate were collected, which represented 18 times the calculated

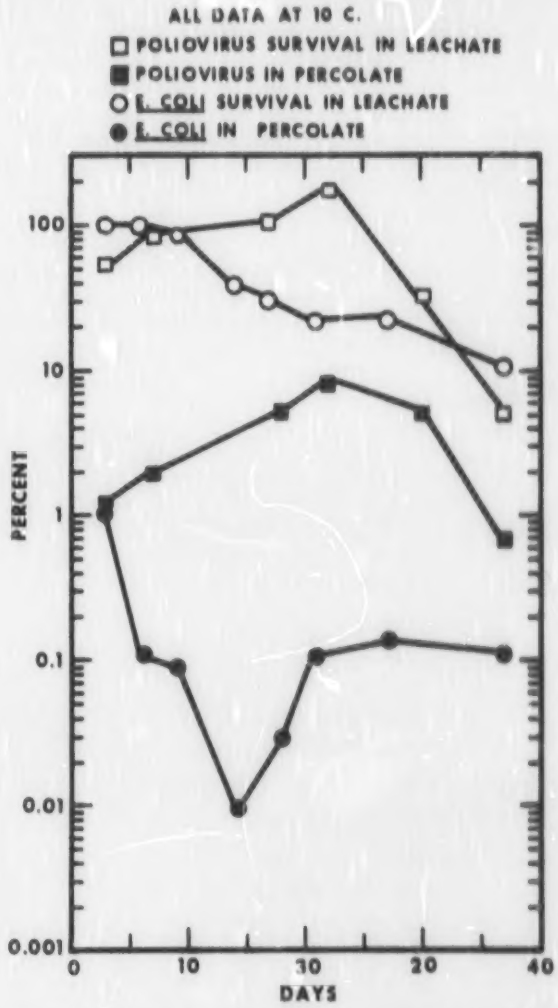

Figure 1. Poliovirus and *E. coli* in municipal landfill leachate and sandy soil column percolates. (All data at 10 C. □ poliovirus survival in leachate; ■ poliovirus in percolate; ○ *E. coli* survival in leachate; ● *E. coli* in percolate.)

pore volume of the soil and 25 times the calculated specific yield of the soil.[6]

From the results of these experiments, it is concluded that uniform soils, particularly soils with high clay content, underlying landfills have the capacity to protect groundwaters from pathogens. This conclusion is limited to continuous strata of soils similar to the types used in this study and is contingent on the distance leachates move through soils. It is, therefore, apparent that landfill site selection is a critical factor in protecting public health.

FUTURE RESEARCH

Additional research is needed to determine a rational assessment of the problem of pathogens in landfills and their migration into the environment by the leaching process. Further research should be undertaken to compare the survival of bacterial indicators of fecal pollution and enteric pathogens. Many methods are available for concentrating viruses in water, and several have been applied to leachates. Comparisons of promising methods could contribute needed information to provide the best detection possible for small quantities of viruses in large quantities of leachates. Microbiological monitoring of leachates and groundwaters below actual landfill sites is desired since there are a limited amount of field data.

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WATER SUPPLY RESEARCH
February 10, 1975

EXAMINATION OF DRINKING WATER SUPPLIES FOR VIRUSES

Walter Jakubowski and Norman A. Clarke*

INTRODUCTION

"What is the potential human health hazard associated with the transmission of viral disease agents by or through drinking water?" This question has been asked for a number of years by those concerned with the treatment and distribution of drinking water. A wide variety of factors must be considered in the course of arriving at a definitive answer.

Human beings are hosts to over 100 members of the enteric virus group (polio-, coxsackie-, echo-, reo-, and adenoviruses). These viruses reproduce in the human digestive tract and are consequently excreted into sewage. From that point of entry into the aquatic environment, their numbers steadily decrease, since viruses cannot reproduce outside a living cell. The action of time, sewage and water treatment processes, sunlight, physical and chemical inactivation by various suspended and dissolved materials, and simple dilution all serve to diminish the concentration of viruses in water. The low concentrations of viruses that conceivably could occur in water, together with the fact that human enteric viruses are transmitted through person-to-person contact, combine to make an assessment of the significance of the water vehicle in the transmission of viruses an extremely difficult problem.

One approach to resolving the question is to determine if viruses can be found in treated drinking water. In the development of virus detection methodology, the seeding of 3.8 liters (1 gal) of water with 4,000 to 40,000 infectious units of virus had been considered, at one time, a "low" virus concentration for laboratory experimentation. Current techniques are able to detect poliovirus at levels of 1 or 2 infectious units per 380 liters (100 gal) when 1900 liters (500 gal) are sampled.[1] However, these methods are not 100 percent efficient in recovering virus in laboratory experiments, and it is therefore not unreasonable to assume that more virus than can be detected may actually be present in field samples. Viruses, in low numbers, have been demonstrated in river water downstream from sewage treatment plant discharges and at drinking water treatment plant intakes.[2]

SURVEYS OF WATER SUPPLIES

Between 1969 and 1971, a pilot study of viruses in water in two Massachusetts communities was conducted, and the first suspected isolation of viruses from treated drinking water in the United States was reported.[3] This report prompted the Water Supply Research Laboratory (WSRL), of the National Environmental Research Center (NERC) in Cincinnati, to conduct a much broader study of the potential problem of viruses in drinking water.

FY 1973 Survey[4]

The study, begun in July 1972, had three main objectives: a) to determine if the results obtained in the pilot study could be confirmed, b) to evaluate three virus concentrating techniques, and c) to relate the presence or absence of viruses in drinking water to treatment processes, water source, and bacterial and zoomicrobial quality. At that time, three WSRL field laboratories participated in the survey: the Northeast (NE), the Gulf Coast (GC), and the Northwest (NW); each lab utilized the virus concentration technique that it had under evaluation. The three techniques had several components in common (Figure 1), but they differed in the type of

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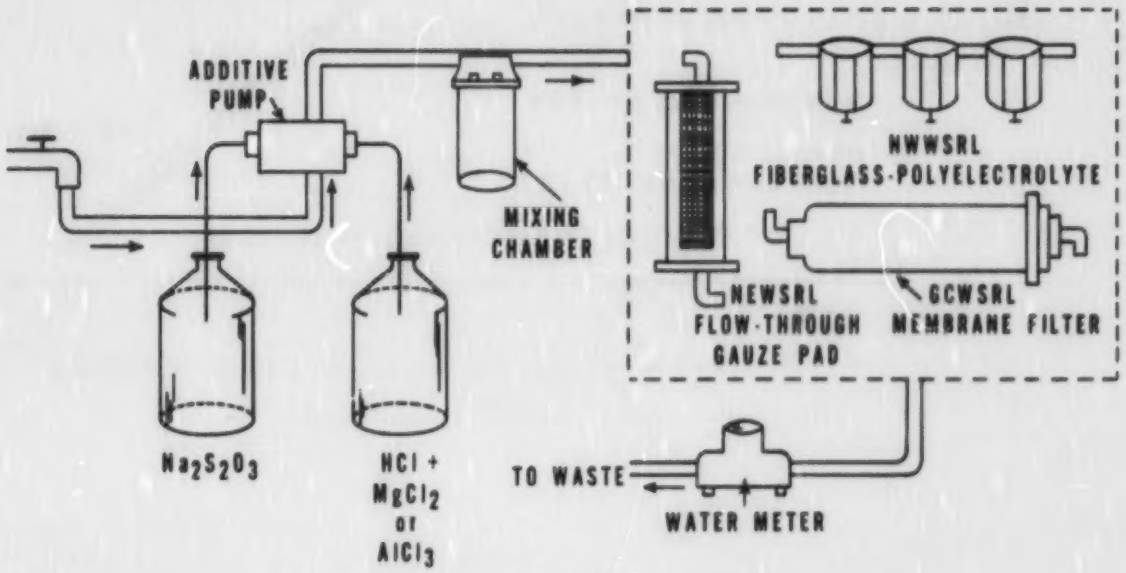

Figure 1. Virus concentrating systems used in FY 1973 survey.

virus adsorbing unit employed: NE used a gauze pad device, GC had a membrane filter, and NW used a fiber glass cartridge coated with an insoluble polyelectrolyte. A hydraulically operated pump[5] was used to introduce sodium thiosulfate (to neutralize chlorine) and hydrochloric acid and magnesium or aluminum chloride (to provide optimum pH and cationic conditions for virus adsorption) via a mixing chamber upstream from the virus retaining unit. Upon return to the laboratory, the sampling devices were put through a procedure designed to release any viruses into a small volume of liquid. These concentrates were then examined for viruses in tissue culture cells and/or mice.

A total of 80 virus study samples were processed from 10 water treatment plants in six States and the District of Columbia. Sample volumes varied from 19 to 950 liters (5 to 250 gal), depending on the technique used and the quality of the water sample. Additionally, 12 positive controls (known virus deliberately added) were processed as a check on the sensitivity of the techniques used. Data indicated that the three techniques had a sensitivity of detecting about one enterovirus unit per 3.8 liters (1 gal) of water when 380 liters (100 gal) or more were sampled.

The GC laboratory examined 45 study samples from two communities in the south and the District of Columbia; the NW laboratory examined 12 samples from two treatment plants in a single community in the west. No viruses were recovered from these samples. Twenty-three study samples were processed from water plants in five different communities in the northeast and the District of Columbia. Only one 19-liter (5-gal) grab sample yielded a virus, which was subsequently confirmed by an independent laboratory as a vaccine-like strain of poliovirus type 3. The sample from which this virus was isolated had been processed early in the course of the study. In addition, this same virus was isolated from a supposedly virus-free control water. At the time, as in the initial phases of any complex undertaking, there were problem areas still unresolved in the processing and examination of the concentrates. There is a definite possibility that this isolate was a laboratory contaminant, and therefore no significance is attached to the isolation.

Coliform organisms were found in only four samples of finished water in the entire study, and these were well within the limit of one coliform/100 ml allowed by the U.S. Public Health Service Drinking Water Standards. Examinations of concentrates for salmonellae, shigellae, and coliforms were negative. Zoomicrobes (nematodes and amoebae) were found in essentially all finished water samples tested.

The 1972-73 study produced valuable background data and uncovered many logistical problems associated with conducting a study of this magnitude. Within the limitations of the methodologies employed, it was concluded that the drinking water from these 10 selected systems was not polluted with sufficient numbers of viruses to be an important vehicle for the transmission of human enteric viruses.

FY 1974 Survey

Six water supplies located in Ohio, Indiana, and Missouri were selected for the survey, and sampling was initiated in October 1973. The sampling device consisted of the additive pump portion of Figure 1 coupled to a 293-mm diameter, disc-type membrane filter holder. The holder contained a stack of three cellulose nitrate membrane filters with porosities of 8, 1, and 0.45 μm. At each sampling site, 380 liters (100 gal) of drinking water were processed through the unit at a flow rate of 3.8 liters (1 gal) per min. After transportation to the laboratory, the filter membranes were treated to elute adsorbed viruses,[6] which resulted in a concentrate volume of approximately 10 ml. The concentrates were examined for viruses in tissue cultures.

Of the 41 samples collected in FY 74, five were samples in which a known virus had been added at random to test the sensitivity of the method in the field. The technicians collecting the samples, processing the eluates, and examining the tissue cultures had no prior knowledge of which samples had received virus. Each of these "positive controls" was a different virus serotype added at a concentration of 100 infectious units per 380 liters (100 gal) of drinking water. Four of the five positive controls were recovered. Failure to recover the fifth control was attributed to difficulty experienced in collection of the sample and equipment failure during the elution process. The 36 test samples have been completely processed through cell cultures, and no viruses have been isolated from these samples.

CURRENT RESEARCH

Field Survey

The virus survey is a continuing task. In July 1974, two major changes were incorporated into the sample collecting procedure: an epoxy — fiber glass filter[7] replaced the cellulose nitrate membranes as the virus adsorbent, and sample volume was increased to 1900 liters (500 gal). The filter apparatus being used in the current survey is shown in Figure 2. Evaluation of this device in our laboratory has revealed a virus-recovery efficiency that is equivalent or superior to that of the membrane filter procedure and allows the processing of larger volumes without clogging difficulties. The same sampling sites are being utilized, although other sites are in the process of being selected.

Methods Evaluation

A close liaison is maintained between the survey group and the virus methods evaluation group. This contact enables improvements uncovered in the laboratory to be more rapidly implemented in the field. Experiments are designed with a view toward application of the results to field survey techniques.

Currently, several virus adsorbents are being evaluated simultaneously in the laboratory under rigorously controlled conditions; these include

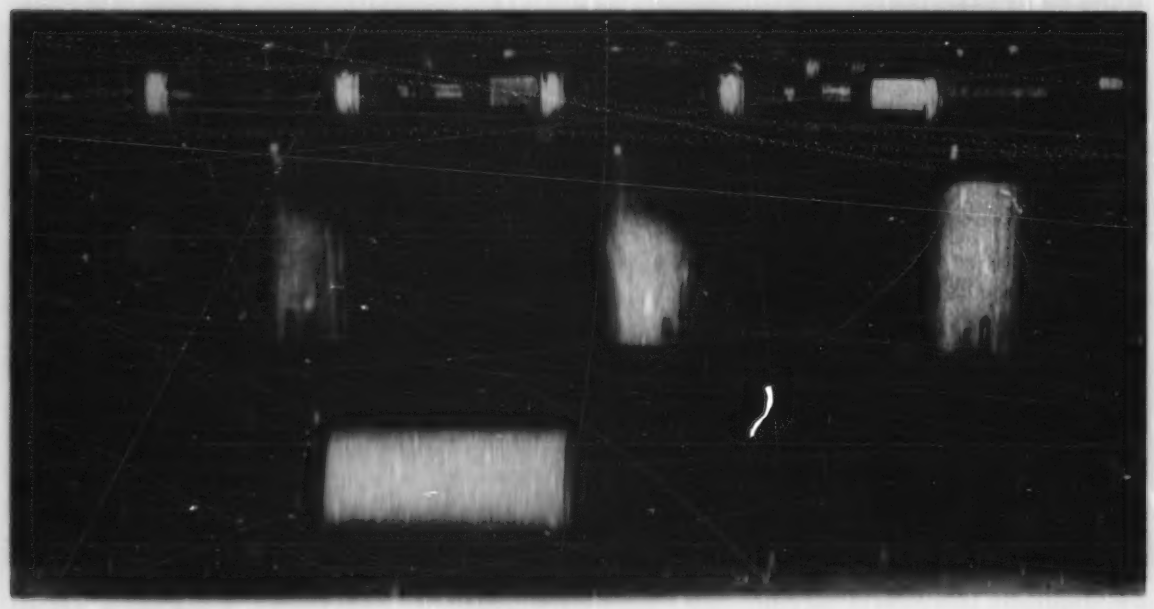

Figure 2. Virus adsorbing filter used in the current survey.

epoxy-coated microfiber glass in disc and cartridge configurations, yarn-wound fiber glass cartridges, and cellulose nitrate membranes. A considerable body of literature exists on the use of these adsorbents for concentrating viruses, but much of this work concerns their use with distilled or deionized water or prefiltered tapwater. Furthermore, unnaturally high levels of virus and low volumes of water were necessarily used in initial evaluations. The efficiency of virus recovery under these conditions can, and indeed does, decrease significantly when 380 liters (100 gal) or more of drinking water, dosed with one virus unit per 3.8 liters (1 gal) or less, is sampled directly from the tap. Consequently, the virus adsorbents are now being compared by WSRL researchers at virus levels that might be expected to occur in natural waters and with sample volumes up to 1900 liters (500 gal) of treated drinking water.

FUTURE RESEARCH

At present, field samples are collected and concentrated onsite and transported to the laboratory for elution and examination. This procedure may decrease the chances of virus isolation because of virus inactivation by adverse conditions during transport. To eliminate this possibility, a mobile, self-contained laboratory is being designed to concentrate and elute samples onsite. The ideal virus concentrator has not yet been developed. Research will continue on improving existing methods and evaluating newer methods as they become available.

The accumulation and analysis of meaningful data are necessary for a valid assessment of the hazard of viral disease transmission through drinking water supplies, and WSRL virologists are directing their efforts to this end.

ACKNOWLEDGMENT

The investigators responsible for conducting the FY 1973 survey at each of the three field laboratories were: Dr. E. W. Akin at Gulf Coast, Dr. O. C. Liu at Northeast, and Dr. J. C. Hoff at Northwest. The FY 1974 survey was directed by Dr. E. W. Akin.

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METHODS DEVELOPMENT AND QUALITY ASSURANCE RESEARCH

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A COPPER-CADMIUM COLUMN FOR MANUALLY DETERMINING NITRATE

M. E. Gales, Jr. and R. L. Booth*

INTRODUCTION

Nitrate has been considered a most important water quality parameter because of its association with human and animal waste. Nitrate is usually the most prevalent form of nitrogen in water, because it is the end product of the aerobic decomposition of organic nitrogen. Nitrate from natural sources is attributed to the oxidation of nitrogen of the air by bacteria and to the decomposition of organic material in the soil. Fertilizers may add nitrate directly to water resources through runoff. Nitrate-rich effluents discharged to receiving streams can produce algal blooms. It has also been recognized that cyanosis due to methemoglobinemia may occur in infants whose drinking or formula water contains a high concentration of nitrates.

The determination of nitrate has always been a difficult problem. Although a number of methods have been developed for its determination, all are subject to some form of interference and none are applicable to a large variety of sample types. Compliance monitoring of point-source discharges, as required by the National Pollution Discharge Elimination System (NPDES), has accelerated the need for a simple, reliable method for nitrate determinations that can be used for all types of water and waste samples.

A number of methods have been developed that are based on reducing nitrate to nitrite; the nitrite (originally present plus reduced nitrate) is determined by diazotizing with sulfanilamide and coupling with N-(1- naphthyl)-ethylenediamine dihydrochloride to form a highly colored azo dye that is measured spectrophotometrically. There are very few known interferences with this colorimetric method, and it has a very high sensitivity.

Since 1951 zinc has been used as a reductant for the determination of nitrate; however, this reaction tends to continue beyond nitrite, and closely controlled conditions are necessary if reproducible results are to be obtained. O'Brien and Fiore[1] automated this procedure, but the efficiency of the reduction varies with time and the number and kinds of samples. Mullin and Riley [2] developed a method that reduces nitrate to nitrite by the use of hydrazine in the presence of copper. The yield of nitrite was not stoichiometric, however, but only 60%. Kamphake *et al.*,[3] in automating this procedure, obtained a 100% reduction of nitrate to nitrite when applying the procedure to surface waters, sewage, and industrial waste. A shortcoming of the copper-hydrazine method, however, is its inability to determine nitrate in saline waters. Cadmium[4] is also used to reduce nitrate to nitrite. The metal can be used as cadmium, amalgam-cadmium, and copperized-cadmium. The automated copper-cadmium method [5] has been used successfully to analyze a variety of water samples.

The manual colorimetric methods most commonly used are the phenoldisulfonic acid, chromatropic acid, and brucine methods. Of this group, the phenoldisulfonic acid method [6,7] is the most popular, but seldom yields reliable results except under carefully controlled conditions. Further, it is subject to interference from chloride and nitrite ions, and the suggested pretreatment for their removal frequently leads to inaccurate results. The chromatropic method[8] is very tedious to apply, and the 2.5-ml sample volume is difficult to pipet accurately. Barium, strontium, lead, iodide, iodate, selenite,

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and selenate interfere with the chromatropic method through the formation of precipitates.

The brucine method, developed by Jenkins and Medsker, [9] gives reliable results for determining nitrate levels in natural waters. In an Analytical Reference Study, [10] the brucine method yielded the best overall results and was recommended for use in the range of 0.1 to 2.0 mg NO_3-N/l. More recently, Holty and Polworowski [11] have shown that the color developed by this procedure does not level off above 2 mg/l (1:1 stoichiometry), but instead drops off to absorbance levels within the range of 0.1 to 2.0 mg of NO_3-N/l of the standard curve before increasing again. When this happens, the concentration of nitrate appears to be within the range of the standard curve, but actually the sample may contain as much as 50 times the apparent concentration of nitrate. Holty eliminated this problem by increasing the concentration of brucine sulfate. Although there are some problems with the brucine method, it has proven to be the most reliable for manually determining nitrate in surface waters and in some industrial wastes. Subsequent studies on a variety of treated waste effluents showed, however, that extremely erratic results were obtained in samples that contained high concentrations of oxidizing or reducing agents and/or high concentrations of organic matter.

The objective of this study was to develop a manual method that could be used to determine nitrate-nitrogen in wide variety of water and waste samples. Based on a literature review, the cadmium reduction method in *Standard Methods*[6] appeared to be most promising. Nitrate is reduced to nitrite in this method by passing the sample through a column containing mercury-cadmium filings. Although this method appears in *Standard Methods*, the interferences are not identified. Changes were made in reagents and sample treatment to make this method identical with the automated copper-cadmium methods. By determining and eliminating the interferences of this method, reliable results were obtained for both water and waste samples over a range of 0.01 to 1.0 mg NO_3-N/l.

EVALUATING AND REVISING THE METHOD

The reduction columns used for this study contained copper-cadmium. Cadmium powder was used in place of cadmium filings. Mercury-cadmium was not used because mercury is considered a hazardous material.

A preliminary evaluation of the manual cadmium reduction method[6] was carried out by analyzing samples of river water, industrial waste, and sewage. The results were compared with those obtained with the brucine method (Table 1). Although both procedures gave comparable results for the river water samples, the brucine method yielded a higher concentration and better recovery of nitrate for the industrial waste sample. On the other hand, with the sewage sample, the cadmium reduction method gave a somewhat higher concentration of nitrate and the brucine method gave better recoveries.

TABLE 1. COMPARISON OF NITRATE RESULTS OBTAINED BY THE BRUCINE AND CADMIUM REDUCTION METHODS[6]

Method	Sample description	NO_3-N, mg/l Present in sample	Added	Found	% Recovery
Brucine	Ohio River	1.1	0.5	1.6	100
Brucine	Industrial waste	16.7	10.0	25.8	97
Brucine	Sewage	0.20	0.50	0.53	75
Cadmium reduction	Ohio River	1.1	0.5	1.6	100
Cadmium reduction	Industrial waste	10.2	10.0	13.0	69
Cadmium reduction	Sewage	0.29	0.50	0.32	40

Additionally, industrial waste samples containing high concentrations of organic matter were also analyzed. In this case, better recoveries were obtained by the cadmium reduction method (Table 2). The cadmium reduction method was also applied to an industrial waste that contained 1.6 mg Fe/l and to a treated waste effluent from the Cincinnati Sewage Treatment Plant. These samples were analyzed by the automated cadmium reduction method and the manual method.[6] Excellent results were obtained by the automated method, but poor results were ob-

TABLE 2. COMPARISON OF THE BRUCINE AND CADMIUM REDUCTION METHODS FOR NITRATES IN SAMPLES CONTAINING HIGH CONCENTRATIONS OF ORGANIC MATTER

Sample	% Recovery: Brucine method	Cadmium reduction method
1	90	91
2	49	25
3	<1	101
4	<1	109
5	54	101

tained by the manual method. As a means of increasing the recovery, the same ratio of sample to ammonium chloride (1:3) that was used in the automated method was applied to the manual method. This modification improved the recovery of nitrate from the industrial waste sample, but results from the sewage sample did not improve.

Subsequently, it was observed that samples containing metals gave low results (Table 3). Apparently, metals precipitated at the top of the column and thereby slowed the flow of the sample. This interference was eliminated by adding disodium ethylenediaminetetraacetate (EDTA)[12] to the ammonium chloride solution. The ammonium chloride-EDTA solution was adjusted to a pH of 8.5 with ammonium hydroxide[13] to hold the EDTA in solution. Use of this reagent complexed the metals present in the sample and thus kept the surface of the cadmium clean. To determine the maximum concentration of EDTA that could be used without affecting the reduction, a series of distilled water solutions containing 1.0 mg of nitrate-nitrogen/l and varying amounts of EDTA were analyzed. The maximum yield of nitrate was obtained with 0.1 gram of EDTA.

TABLE 3. THE EFFECT OF EDTA ON THE RECOVERY OF NITRATE FROM DISTILLED WATER

EDTA, conc. g/80 ml	NO_3-N in standard, mg/l	NO_3-N found, mg/l
0.1	1.0	0.99
0.2	1.0	0.93
0.3	1.0	0.91

The use of EDTA produced excellent nitrate recovery from surface water (Table 4), but recovery from sewage still did not improve. Suspecting that oil and grease present in domestic waste could be interfering by coating the cadmium and thereby reducing its efficiency, the column was washed with acetone and, indeed, large amounts of oil and grease were removed. Accordingly the oil and grease were removed from the sample prior to its passage through the column by adjusting the pH of 100 ml of sewage to 2 and extracting it with two 25-ml portions of Freon. After the extraction, the pH of the sample was readjusted to 6. With this pretreatment of the sample, 96% recovery of nitrate was obtained from the sewage samples, and the column was used to efficiently analyze 48 sewage samples.

After the above modifications were made, the percent recovery of nitrate from industrial waste, sewage, river water, tap water, and ocean water was determined. As shown in Table 5, excellent results were obtained for all samples. The percent recovery was verified by spiking a sewage sample at three levels — 0.2, 0.5, and 1.0 mg of NO_3-N/l. The recovery of these spikes was 100%, 102%, and 101%, respectively.

TABLE 4. THE EFFECT OF EDTA ON THE RECOVERY OF NITRATE FROM SURFACE WATER

Sample	NO_3-N in sample, mg/l	NO_3-N added, mg/l	% Recovery
Little Miami River	0.71	0.50	97
Ohio River	1.7	1.0	99
Muddy Creek	0.42	0.50	99

The precision of this method was also determined by analyzing four concentration levels of nitrate in sewage. They included a low concentration near the detection limit of the method, two intermediate levels, and one level at the upper limit. The concentrations of nitrogen used in this study were 0.04, 0.24, 0.55, and 1.05 mg/l. The standard deviations were ±0.005, ±0.004, ±0.005, and ±0.01 mg/l, respectively.

APPLYING THE METHOD

A very simple and accurate method for determining nitrate in all types of water and waste samples has been developed. Interference from metals is eliminated with the use of EDTA; oil and grease are removed by acid extraction with Freon. The sensitivity of this method is 0.01 mg NO_3-N/l, with a working range of 0.01 to 1.0 mg N/l. Essentially, a filtered sample is passed through a column contain-

TABLE 5. RECOVERY OF NITRATE FROM A VARIETY OF SAMPLES

Sample	NO_3-N in sample, mg/l	NO_3-N added, mg/l	NO_3-N found, mg/l	% Recovery
Industrial waste	23.0	20.0	44.0	102
Sewage	0.01	0.50	0.48	94
Little Miami River	0.71	0.50	1.2	98
Muddy Creek	0.42	0.50	0.91	99
Ocean water	0.08	0.50	0.58	100
Tap water	1.5	0.50	2.0	100

ing granulated cadmium-copper to reduce nitrate to nitrite. Separate (rather than combined) nitrate-nitrite values are readily obtained by carrying out the procedure first with, and then without, the initial Cd-Cu reduction step. This method will be included in the U.S. Environmental Protection Agency's 1974 edition of "Methods for Chemical Analysis of Water and Wastes."[14]

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ADVANCED WASTE TREATMENT RESEARCH

April 11, 1975

POLLUTION FROM URBAN LAND RUNOFF

Anthony N. Tafuri*

INTRODUCTION

The most obvious, easily recognizable sources of potential water pollution are domestic and industrial wastes. Inadequately treated discharges from municipal and industrial sewers (point sources) pose the greatest single threat to water quality. Consequently, point sources have long been studied, and much has been learned about how to diminish or eliminate their influence on water quality.

The cost of waste management increases rapidly as higher levels of treatment are required, and the ultimate national goal of "zero discharge" is expected to require public investment many times that of present levels. Even with "zero discharge" of municipal and industrial wastes, however, national goals for water quality may not be achieved in some areas because of pollution from urban land runoff and other nonpoint sources. A recent report of the Council on Environmental Quality[1] indicated that in 80 percent of the urban areas studied, downstream water quality was not controlled by point sources.

DESCRIPTION OF URBAN LAND RUNOFF

Urban land runoff is the discharge of excess urban surface waters, occurring as a direct result of precipitation, into the nearest natural or manmade collection channel. Urban runoff has been found to contain significant amounts of organic wastes, nutrients, fecal bacteria, heavy metals, and solids. As the rainwater passes through the atmosphere, it sweeps out particulate matter, gases, and odors produced by natural conditions and industries. From there, it flushes the earth's surface, washing all objects with which it comes in contact. Materials can be dissolved, suspended, floated, or forcefully removed and carried away. Essentially, what happens is that "Mother Nature" cleanses the urban environment in a fashion similar to washing off the patio with a hose.

In light of the types of substances ready for water transport in the urban environment, it is not surprising that urban runoff contains substantial quantities of objectionable materials. Fertilizer, animal wastes, leaves, and grass clippings are typically found on lawns and other surfaces. Streets and parking lots contain engine oil, combustion byproducts, and miscellaneous litter. The materials worn from tires and brake shoes are also distributed in the streets. Heavy particulate matter in airborne waste settles out and is transformed into water pollutants. Litter such as cans, bottles, plastic, and paper products can be washed away. Erosion and sedimentation also contribute pollutants to urban runoff.

The seriousness of urban runoff has, to some extent, been overshadowed by the urgent need for treatment of domestic and industrial wastes. Consequently, runoff has not been adequately studied, and its role in water pollution control is poorly understood. Communities may make expensive sewage treatment plant improvements and still not achieve water quality goals because of the impact of urban runoff or some other nonpoint source. Steps must be taken to harmonize land use with water quality or to treat the runoff waters to reduce their adverse effects on receiving streams.

DURHAM RESEARCH

In a study by Dr. Edward H. Bryan[2] urban runoff was measured in Durham, North Carolina, from August 1, 1968 through June 30, 1970. Dr. Bryan found that with respect to biochemical oxygen demand (BOD), the total weight contribution of urban runoff from the study basin was estimated to equal that of sanitary wastewater effluent following secondary treatment. The contribution of chemical oxygen de-

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mand (COD) in the stormwater was greater than that attributable to discharge of raw sanitary wastewater from a strictly residential or average urban area. The total solids were found to be substantially larger than would be expected from average raw domestic wastewater. The contribution of phosphate was found to be nominal for urban stormwater when compared with that of domestic wastewater.

Follow-Up Study

In an attempt to further characterize pollution from urban runoff, a followup study of the Durham watershed was initiated in 1971 under the joint sponsorship of the U.S. Environmental Protection Agency and the North Carolina Water Resources Research Institute.[3] The study site was the upper Third Fork Drainage Basin in Durham, North Carolina — a 1.67-square-mile drainage basin completely within the city limits — with its northern boundary located in the heart of the downtown business district.

The primary objectives of the Durham study were characterizing urban runoff with respect to pollutant concentrations and comparing the impact of urban runoff's annual yield of pollutants on water quality with that of domestic wastes. The characterization of urban runoff is important, since planners, engineers, and other interested groups need to know what pollutants are associated with this source, how much is contributed during specified time periods, and how pollutants are distributed within each runoff event. Careful evaluation of all sources of potential water pollution is essential for public officials who must decide how to spend the limited funds available for pollution control.

Research Findings

The average concentration and annual yield of pollutants in runoff from the Durham Third Fork Creek study area are given in Table 1. These data are in general agreement with the values obtained by Bryan[2] and investigators in other metropolitan areas.

The Third Fork Creek study area is served by the Durham Third Fork Sewage Treatment Plant, which removes an average of 91 percent of the organics (as measured by BOD) and 85 percent of the suspended solids. The annual yield (1972) of raw and treated municipal waste from the Durham Third Fork Sewage Treatment Plant is compared with annual yield from urban land runoff in Figure 1. From this figure it can be seen that after providing secondary treatment of municipal wastes, the largest single source of pollution on an annual basis from the watershed is urban runoff. In fact, the dissolved oxygen content of the receiving watercourse during storm flows was independent of the degree of treatment of municipal wastes beyond secondary treatment. Even when the

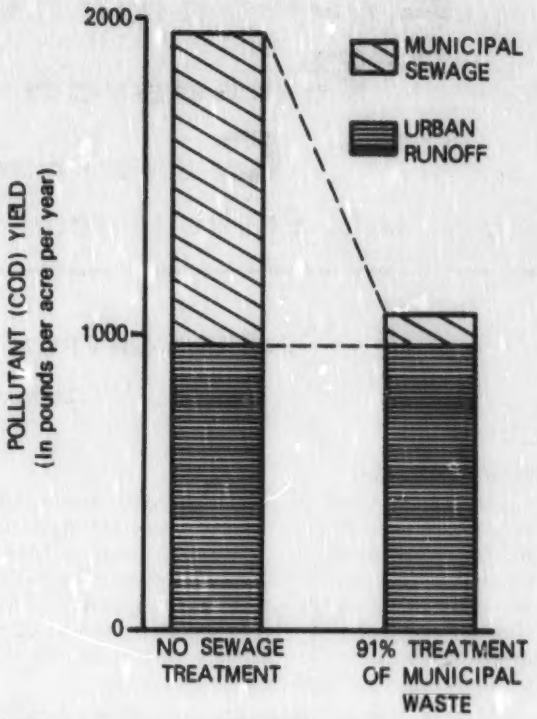

Figure 1. Annual pollutant (COD) yield from urban land runoff. (To convert pollutant yield to grams per year per square meter, multiply by 40.88.)

secondary plant was assumed to be upgraded to "zero discharge," oxygen sag estimates were unchanged. The results from this figure cannot be assumed to represent precisely the severity of the runoff problem since runoff is often accompanied by high flow rates (and therefore high degrees of dilution) in receiving waters. It can be seen, however, that the runoff problem can be a very important one, especially when receiving water flow is low.

Another project objective was to investigate the applicability and effectiveness of chemical coagulation and sedimentation of urban land runoff. Sedimentation for 15 minutes under ideal conditions was found to remove an average of 61 percent of the COD, 77 percent of the suspended solids, and 53 percent of the turbidity. Alum, with or without coagulant aids, was judged to be the most effective coagulant for chemical treatment of urban land runoff. An average of 60 mg alum per liter was found to effect removals of COD, suspended solids, and turbidity of 84, 97, and 94 percent, respectively. Sedimentation, which is much less costly than chemical coagulation, removed a significant portion of organics and solids and should be considered as the first alternative in treatment of urban runoff. Alum

TABLE 1. AVERAGE CONCENTRATION AND ANNUAL YIELD OF POLLUTANTS FROM THE DURHAM STUDY AREA

Pollutant	Average concentration mg/l*	Annual yield lb/acre/yr.†
Chemical oxygen demand	170	938
Total organic carbon	42	187
Total solids	1440	7700
Total susp. solids	1220	6700
Kjeldahl nitrogen as N	1.0	6
Total phosphorous as P	0.8	5
Aluminum	16	64
Cobalt	.16	2
Chromium	.23	1.6
Copper	.15	1.6
Iron	12	102
Lead	.46	71
Manganese	.67	1.2
Nickel	.15	2.9
Zinc	.36	2
Fecal coliforms (No./ml)	230	—

*Except coliforms

†To convert to grams/yr./sq. meter multiply by 40.88

produces significant increases in pollutant removal over sedimentation.

The effect of sedimentation and/or chemical treatment of urban runoff in reducing the total annual pollutant yield from an urban area is shown in Figure 2 for three different options: (1) no sewage or urban runoff treatment, (2) 91 percent sewage treatment plus sedimentation of urban runoff, and (3) 91 percent sewage treatment plus chemical treatment of urban runoff. Figure 2 illustrates that treatment of both domestic wastes and nonpoint sources may be necessary to substantially reduce pollution discharges in urban areas.

SUMMARY AND CONCLUSIONS

Urban land runoff is a significant source of pollution. When compared to the raw municipal waste generated within the study area, the annual urban runoff of COD was equal to 91 percent of the raw sewage yield, the BOD yield was equal to 67 percent, and the suspended solids yield was 20 times that contained in the raw municipal wastes.

If Durham provided 100 percent removal of organics and suspended solids from the raw municipal waste on an annual basis, the total reduction of pollutants discharged to the receiving water would only be 52 percent of the COD, 59 percent of the ultimate BOD, and 5 percent of the suspended solids.

During storm flows, dissolved oxygen content of the receiving watercourse was found to be independent of the degree of treatment of municipal wastes beyond secondary treatment. Oxygen sag estimates were unchanged even if the secondary plant was assumed upgraded to zero discharge.

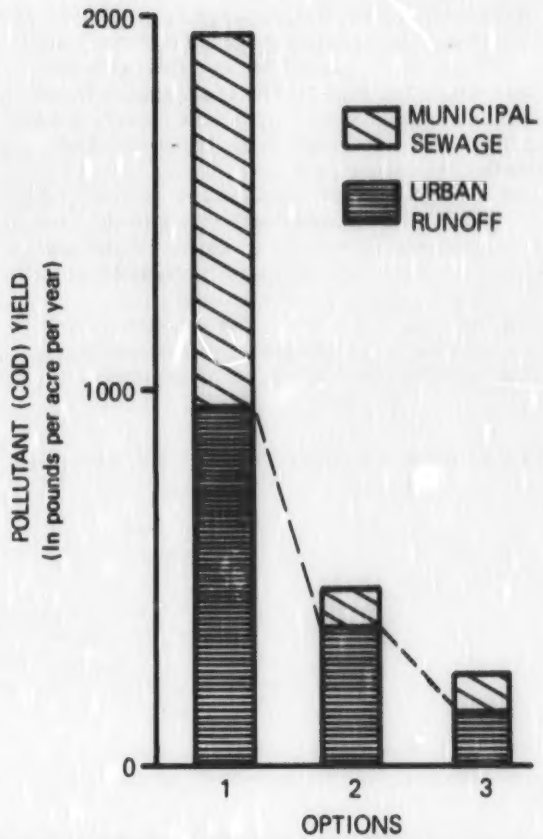

Figure 2. Pollutant (COD) yields with different treatment options: (1) no sewage or urban runoff treatment; (2) 91 percent sewage treatment plus sedimentation of urban runoff; and (3) 91 percent sewage treatment plus chemical treatment of urban runoff.

Solid wastes such as beer cans, broken glass bottles, garbage, bed springs, shopping carts, etc., also find their way into urban stream beds. These solid wastes, believed to be typical of urban streams, not only contribute to lower water quality, but are aesthetic pollutants that add to property devaluation and are a hazard to public safety as well.

In urban drainage basins, the beneficial effects of upgrading secondary municipal waste treatment plants may be uncertain in view of the apparent relative impact of urban land runoff on receiving water quality.

Consequently, urban areas planning to upgrade secondary sewage treatment plants because of pos-

sible contravention of stream standards should carefully assess the pollution potential of urban runoff.

Full-scale evaluations of the efficiency, economics, and applicability of alternative facilities to reduce the impact of urban land runoff on water quality should be made. Some handling techniques worth considering are:

- Storage of urban runoff during periods of high flows, with eventual pumpback to the sewage treatment plants during periods of low sewage flow. The excess surface water may be stored in tanks, ponds, or underground.
- Physical solids separation by such devices as fine mesh screens and swirl concentrators.
- Separate physical/chemical treatment.
- Plain sedimentation through manmade or natural settling basins, with chlorination where necessary.

Citizens can also be encouraged to help improve the quality of urban streams by avoiding littering and supporting local solid waste disposal programs.

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*National Technical Information Service, Department of Commerce, Springfield, Virginia 22151.

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April 25, 1975

STRIVING FOR ZERO DISCHARGE OF INDUSTRIAL WASTES

J. Ciancia and H. M. Freeman*

INTRODUCTION

Recent Federal legislation (PL 92-500) declares that it is the national goal to eliminate the discharge of pollutants into the navigable waters of the United States by 1985. As a result, the closed-loop approach to pollution control has assumed a dominant role in the industrial research and development program of the U.S. Environmental Protection Agency. The elimination of pollutant discharges can be accomplished by chemical recovery and water reuse at the same manufacturing facility or through by-product recovery of the waste materials.

However, the concept of recovering usable constituents from wastes and recycling treated wastewaters generally requires considerable ingenuity and a thorough insight into the processing operations. Many technical and economic barriers must be overcome before this concept becomes a reality, particularly when the processing operation generates a complex waste containing a variety of contaminants. The EPA industrial research, development, and demonstration program, by advancing technically sound and economically feasible improvements in waste treatment through the development and demonstration stages, is attempting to shorten the time between the research and implementation stages of industrial closed-loop control. The Industrial Waste Treatment Research Laboratory, part of the National Environmental Research Center in Cincinnati (NERC-Cincinnati), is the EPA laboratory designated to carry out this industrial research and development program.

This article briefly describes a number of projects in the metal finishing and nonferrous metals industries in which closed-loop-type technology has been developed or demonstrated. To date, EPA has supported over 30 research and development studies in these industries. Technological developments for recovering chemicals and reusing wastewater from other industries will be discussed in future NERC-Cincinnati newsletters.

Closed-loop-type treatment of wastes from the metal finishing and nonferrous metals industries not only eliminates the discharge of pollutants into the environment but also conserves natural resources and prevents the buildup of solid wastes (sludges) produced by chemical treatment techniques.

SIGNIFICANCE OF METAL FINISHING AND NONFERROUS METAL POLLUTION PROBLEMS

The metal finishing and nonferrous metals industries produce a multitude of waste discharges ranging from very dilute large flows to highly concentrated batch dumps.[1-10] The major contaminants in the wastes from these industries are a wide variety of toxic heavy metals, cyanide, fluoride, acid or alkalies, oils, and suspended solids. The wastes produced in the metal finishing industry come mainly from dumped spent-processing baths, rinsewaters used to wash off process solution adhering to the surface or trapped in crevices of parts removed from metal finishing baths, and accidental spills and leaks. The major wastewaters arising in the nonferrous metals industries are mud slurries, flotation plant tailings, gas scrubber waters from smelting and refining, discarded leach liquors, spent electrolytic solutions, clean-up water, slag and ore storage runoff, direct contact cooling water, and metal fabricating processing bath dumps, rinsewaters, and leaks and spills.

The large amounts of toxic and other harmful contaminants in metal finishing and nonferrous metal

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industry wastes represent a serious pollution problem. The wastes are harmful to humans, animals, and various forms of aquatic life; and, if discharged into sewers, can vitiate the biological treatment processes and cause corrosion of the metal and concrete structures.

CLOSED-LOOP TREATMENT PROCESSES

Reverse Osmosis

Under an EPA grant to the American Electroplaters' Society (AES), Abcor, Inc. (Cambridge, Massachusetts) conducted a project to demonstrate the technical feasibility and determine the economics of reverse osmosis (RO) for treating metal finishing rinsewaters by recovery of the chemicals and reuse of the water.[8]

The application of RO for closing the loop on metal finishing rinse wastes is shown in Figure 1. Rinsewater from the first tank, which would otherwise be discharged to the drain, is separated by the RO system into a "concentrate" and "permeate" stream. The concentrate stream is returned to the processing bath, and the purified or permeate stream is recycled to the rinsing operation. A small evaporator is required for those metal finishing operations where insufficient natural evaporation occurs in the bath.

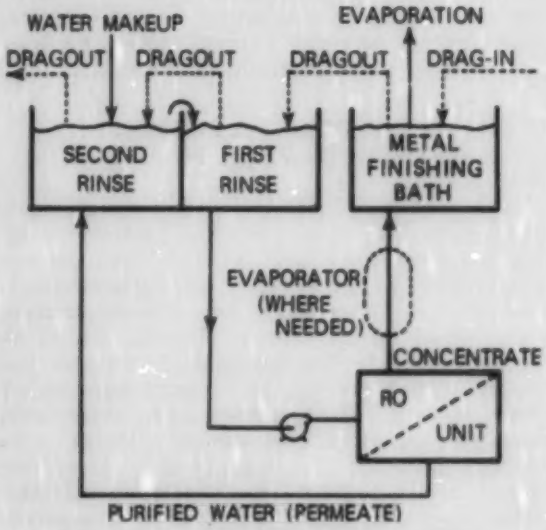

Figure 1. Reverse osmosis recovery system.

The major advantages of RO over other metal finishing rinse recovery processes are low capital and energy costs, simplicity of operation, and compact equipment requiring a minimum of space. The modular nature of RO units makes them particularly well suited for small-scale installations. The most important limitations of RO are the pH and operating temperature limitations of presently available commercial membranes and the inability of RO to concentrate the chemicals in the rinsewaters sufficiently for certain baths to close the loop without a small amount of additional concentration by an evaporator.

The first phase of the AES study involved the testing of all commercially available membranes in modular pilot plant equipment on actual metal finishing rinsewaters. Tests conducted on cellulose acetate tubular, cellulose acetate spiral wound, and hollow fiber polyamide membranes showed that RO is presently an attractive method for treating mixed plant effluents and rinsewaters from Watts nickel, nickel sulfamate, copper sulfate (acid copper), copper pyrophosphate, nickel fluoroborate, and zinc chloride baths, as well as copper, zinc, and cadmium cyanide baths. However, the technique was found to be unsuitable for treating chromic acid and very high or low pH rinse wastes. After short-range testing, several new membranes now under development appear promising for expanding the pH range and capabilities of RO and for handling oxidizing chemicals such as chromic acid.

The hollow fiber polyamide membrane had a long operating life (3 years) over the pH range of 4 to 11. In contrast, the cellulose acetate spiral wound and tubular membranes had reasonable operating lives over a pH range of 2.5 to 7. Thus, the polyamide hollow fibers appear to be the only attractive membrane for alkaline cyanide and noncyanide rinsewaters at present, and probably should be satisfactory for most alkaline applications except where very high pH levels result from the required concentration. On the other hand, the cellulose acetate membranes are more suitable for the acid baths and should be effective in treating these rinsewaters except where the pH is extremely low or oxidizing acids are present.

The second phase of this AES project is a demonstration study of a full-scale, polyamide hollow fiber RO system for treating rinsewater from a copper cyanide plating bath in a plating shop. Although the RO system only concentrates the copper cyanide in the rinsewater to a fraction of bath strength, copper cyanide plating solutions are operated hot, and there is sufficient loss of water from the process solution to permit return of all the chemicals to the bath without the need for an evaporator. After satisfactory initial operation, the company changed the copper cyanide processing operation and the RO system had to be moved to another plant, delaying the completion of the project.

Process Changes and Electrolytic Recovery

One of the earlier projects supported by EPA was a full-scale demonstration at Volco Brass & Copper

Company (Kenilworth, New Jersey) to evaluate the effectiveness and economics of a pollution control system that involves process changes, electrolytic copper recovery, and integrated rinse treatment of brass mill wastes.[2]

The plant produces about 75 tons of typical copper and cuprous alloy wire per day and uses a hot sulfuric acid primary pickle followed by an ammonium bifluoride — chromic acid secondary pickle. Waste treatment was needed for both the spent pickling solutions and the dilute wastes from the rinsing operations following the various cleaning steps. Conventional chemical treatment of these wastes would result in relatively high capital and operating costs, large-scale equipment, and substantial amounts of potentially toxic sludge.

The new approach demonstrated under the grant involves: the installation of an electrolytic recovery system in the primary pickle bath to remove copper, which would otherwise build up requiring the sulfuric acid solution to be dumped; the replacement of the chromic acid — ammonium bifluoride secondary pickle with a hydrogen peroxide — sulfuric acid solution containing proprietary additives; and the incorporation of an integrated chemical rinse system. The hydrogen peroxide oxidizing treatment in the secondary pickling bath results in the formation of water and cupric sulfate buildup, which is periodically removed by simple crystallization and added to the electrolytic recovery system or sold separately as a byproduct. The integrated chemical rinse is a recycle system in which the copper sulfate washed from the wire is precipitated by the alkalinity in the rinsewater and settled as a salable dense copper sludge in a reservoir tank.

Therefore, the new pollution abatement approach eliminates such troublesome contaminants as chromium, ammonium, and fluoride; recovers essentially all of the copper; prevents the dumping of the process baths; and reduces the plant's process water requirements by about 90 percent. The company estimates that when the cost of pickling plus conventional treatment is compared with that of pickling plus the new waste treatment /recovery system, a substantial savings results — $540 daily compared with $194 daily. In addition, the new system requires much less floor space and eliminates a troublesome sludge problem.

Electrodialysis

A grant was awarded to the RAI Corporation (Long Island, New York) to demonstrate (in prototype equipment treating a simulated waste) the feasibility of using electrodialysis to close the loop on copper cyanide rinsewater.[3]

The application of the electrodialysis process is shown in Figure 2. The system is designed to main-

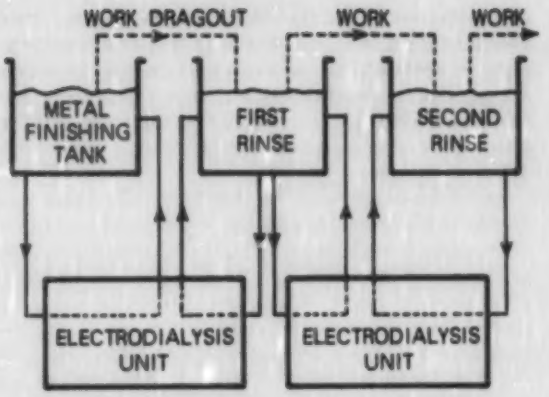

Figure 2. Electrodialysis recovery system.

tain the required quality of water in each rinse tank by returning the amount of chemicals carried over with the parts back to the preceding tank, which contains a much higher concentration of chemicals. The ionic species in the feed entering each electrodialysis stack are transported through the membranes by the application of direct current. As the rinsewater flows through every other compartment, the cation exchange membrane permits the passage of cations and rejects anions and the anion membrane allows the passage of anions and rejects cations. Thus, the rinsewater flowing through these compartments is purified and a concentrate is formed in the adjacent compartments.

The study showed that the chemicals in the rinsewater could be concentrated to slightly over 70 percent of bath strength. For copper cyanide plating, this level of concentration is sufficient to return all of the chemicals to the processing operation since the bath is operated hot and a significant amount of natural evaporation occurs. The ability of electrodialysis to achieve high levels of chemical concentrations should permit the process to economically close the loop on a large number of rinse wastes without the need for additional evaporation.

A grant was recently awarded to establish the effectiveness and economics of the electrodialysis process for closing the loop on a brass cyanide rinse waste under actual plant conditions with full-scale equipment.

Ion Exchange

A unique ion exchange process has recently been developed for recovering phosphoric acid used in the bright finishing of aluminum parts; it also has application for purifying other processing acids used in the nonferrous metal and metal finishing industries.

The process was investigated through the pilot plant stage by Lancy Laboratories (Zelienople,

Pennsylvania). On the basis of this work, EPA awarded a grant to the Douglas & Lomason Company to optimize the new ion exchange system (under pilot plant conditions) and to evaluate the effectiveness and economics of the process for treating phosphoric acid wastes produced in their Cleveland, Mississippi, plant.[7]

Referred to as "acid retardation," the technique employs a strongly basic anion exchange resin for separating the phosphoric acid and aluminum contamination that builds up during the bright finishing of aluminum parts. The anion exchange resin accomplishes the separation by retarding the phosphoric acid as the processing solution flows through the bed. The aluminum remains in the solution and passes out of the column into the effluent. The acid is then eluted from the bed with water, eliminating the use of the chemicals that are needed to regenerate the resin in conventional ion exchange systems.

The completed pilot plant study shows that over 75 percent of the phosphoric acid can be recovered for reuse in the processing operations and that the technique is more economically attractive than alternative methods for treating or disposing of the spent phosphoric acid. The full-scale demonstration study has recently been initiated to evaluate the ion exchange process under actual plant conditions.

CONCLUSION

By continuing to establish and support research programs involving innovative technology such as described herein, the Industrial Waste Treatment Research Laboratory of NERC-Cincinnati intends to play a major role in attaining the national goal of zero discharge of industrial wastes.

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SOLID AND HAZARDOUS WASTE RESEARCH LABORATORY

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SEWER TRANSPORT OF HOUSEHOLD REFUSE A REPLACEMENT FOR THE REFUSE TRUCK

Oscar W. Albrecht and Donald A. Oberacker*

BACKGROUND

Today's modern sewer networks evolved from yesterday's cesspools and open trenches in the interest of health, sanitation, and safety. Then in the 1950's, the concept of wet-transporting putrescible kitchen wastes was introduced and subsequently practiced by way of the now well-known home garbage grinder. The next logical step could be to grind and mix the remaining household wastes with sewage — a procedure referred to as the wet-transport system. The thought of putting ground-up refuse into sewers is perhaps disturbing, but so was the idea of the kitchen garbage disposal when it was first introduced. Recent studies have shown that the concept might have promise in the future.

Recently, a comprehensive analysis was made of the wet transport system to determine its technological and economical feasibility.[1] Previous studies had investigated mainly the technical aspects of wet transport systems, with very little attention given to economic feasibility.[2-4]

The recent Curran study suggested that the wet transport system would be technologically feasible and convenient, but more expensive than the conventional method of collecting and transporting refuse. A cost analysis of the two systems (based on 1973 prices) indicated that the wet transport system would probably cost each household between $113 and $196 annually, depending on population served and refuse generation rate, compared with $43 to $115 annually for the present conventional system (Table 1). A comparison of these costs suggests that the wet transport system is uneconomical for present wide-scale application. The concept may, however, see gradual implementtion through its appeal to certain consumers influenced by its convenience, aesthetics, and promotion by manufacturers and residential developers.

MECHANICS OF WET TRANSPORT SYSTEMS

A considerable amount of research has been conducted on finding alternative systems for grinding household refuse, feeding the refuse into the sewer

TABLE 1. COMPARATIVE ANNUAL COSTS PER HOUSEHOLD FOR CONVENTIONAL AND WET TRANSPORT SYSTEMS, BY SIZE OF POPULATION AND REFUSE GENERATION RATE*

Item	Total system cost range	
	Conventional sytem†	Wet system
Population size‡		
50,000	$57-115	$132-196
100,000	52-110	126-184
500,000	43-101	113-160
Refuse generation rates§		
1.02 kg	43-101	113-160
1.53 kg	45-103	126-172
2.04 kg	47-105	134-181

*Source: Curran Associates, "A Preliminary Assessment of Wet Systems for Residential Refuse Collection," EPA Contract No. 68-03-0183 (March, 1974), p. 152.

†Includes the usual costs for sewage collection and sludge handling and assumes incineration of refuse.

‡A generation rate of 1.02 kilograms per capita per day is assumed.

§Kilograms per capita per day are for a population of 500,000.

*Mr. Albrecht is a research economist and Mr. Oberacker is a mechanical engineer with the Solid and Hazardous Waste Research Laboratory, National Environmental Research Center — Cincinnati (513-684-4484). Acknowledgement is made to Dr. H. E. Bostian and Dr. H. C. Goddard for providing helpful comments and review. Dr. Bostian served as co-project officer along with Mr. Oberacker for the Curran research project.

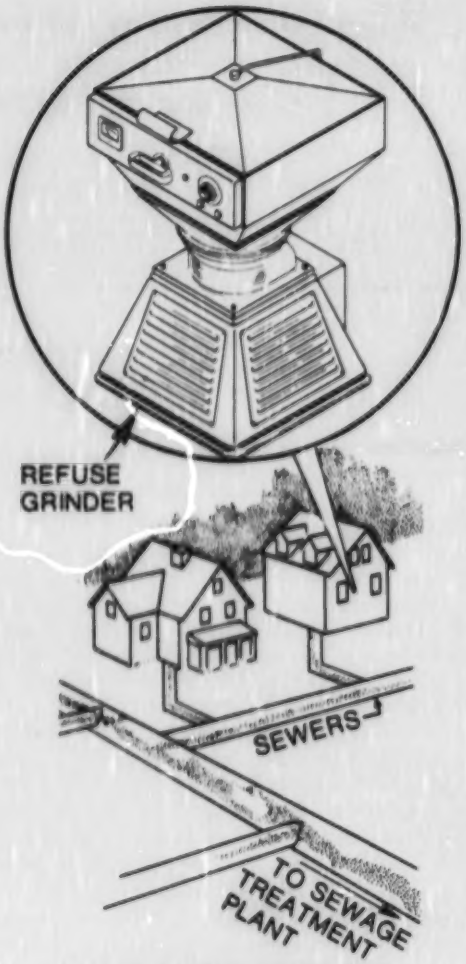
along with ordinary sewage, determining whether the mixture flows well through the system, and determining the proper treatment and disposal. The studies referred to above included investigations of both individual and multi-household or apartment grinders, plumbing requirements, new versus old sewers, rheological properties, and necessary modifications to existing sewage treatment plants. Some household refuse such as yard waste, discarded bicycles, TV sets, or large metal cans, etc., are too bulky or difficult to grind in household refuse grinders. These nongrindable items would have to be collected at intervals by conventional methods. Small metal cans, bottles, paper, and bones could be ground and discharged to the sewer along with the putrescible garbage. In older, slower-flowing sewers, metals and glass may have to be collected separately because of settling problems.

COMPARATIVE COSTS OF WET TRANSPORT AND CONVENTIONAL SYSTEMS

Expenditures for collecting household refuse have increased rapidly in recent years. State and local per capita expenses for solid waste management rose from $3.69 in 1962 to $6.98 in 1971, a 90 percent increase. These figures do not reflect social costs in terms of air, water, land, and noise pollution, disease vectors, etc. Nor is the value of household labor associated with handling residential refuse included in these figures. Much of the increase in expenditures is due to labor costs. Between 1962 and 1971, the annual wage increases to private refuse collection and disposal workers ranged between 3.4 and 7.2 percent in several major cities.[5] Recent wage increases to refuse workers have outpaced the general inflationary trend.[6] The trend in wage rates, coupled with increased unionization and the potential for strikes, means that labor costs will continue to significantly influence refuse collection costs.

Some savings in conventional collection systems can be effected by substituting capital for labor inputs — the use of larger trucks and transfer stations, for example. There are limits to this substitution, of course. The use of capital in place of labor also involves an opportunity cost which must be considered, regardless of whether the collection is operated by a private firm or by local government. Current interest rates are near record highs, and refuse equipment costs have also risen (though not as rapidly as wages). Another factor in the higher cost of solid waste management is the steady increase of land values, particularly near major cities where land available for refuse disposal is at a premium.

The total system costs for the wet transport system and the conventional method of collection and transport on a generalized basis, by size of population area and refuse generation rate, can be compared in Table 1. The Springfield, Massachusetts,



A household refuse grinding system.

area was selected to represent costs for a typical standard metropolitan statistical area (SMSA) (Table 2). For the costs in Table 1, it was assumed that the refuse was incinerated; Table 2 reflects disposal by sanitary landfill. Generally, landfill is a less expensive method of disposal. Economy of scale was shown with respect to size of population served, irrespective of the system, but costs for both methods increased with higher per capita refuse generation rates.

In Table 2, the estimates of component costs for the Springfield SMSA produced some interesting observations. Collection costs for nongrindables would be only $2.50 annually per household, compared with $17.50 by the conventional system. On the other hand, sewer maintenance cost would be at least $6.00, compared with the present $2.00. Total system costs from between $114.50 and $142.50 for

TABLE 2. COMPARATIVE ANNUAL COSTS PER HOUSEHOLD FOR CONVENTIONAL AND WET TRANSPORT SYSTEMS, SPRINGFIELD SMSA*

Item	Cost	
	Conventional system	Wet transport system
Refuse collection	$17.50	$2.50†
Bulky refuse collection	1.00	2.00
Disposal‡	3.50	1.00
Sewer maintenance	2.00	6.00-9.00§
Grinding	—	80.00-105.00
Treatment	9.00	12.50
Sludge handling	5.50	8.50
Sludge disposal	0.75	2.00
Annual Total	39.25	114.50-142.50π

*Source: Curran Associates, "An Assessment of Wet Systems for Residential Refuse Collection: Summary Report," EPA Contract No. 68-03-0183 (March, 1974), p. 80. Data are 1973 price estimates for Springfield, Massachusetts.

†Collection of nongrindables only.

‡Assumes landfill of refuse.

§Assumes one flushing per year. If four flushings are required, the cost would rise to $24 to $36.

πExcluding the grinder system costs, total costs would be $34.50 to $37.50.

the wet transport system are thus significantly higher than the $39.25 for the conventional system.

Specific variables found to influence the cost of wet transport were:

- Population density (single-family versus multi-family dwellings)
- Refuse generation rate per capita
- Age of housing unit
- Cost of grinder (including plumbing, maintenance, power, etc.)
- Proportion of nongrindables
- Water consumption
- Hydraulic transport (including sewer modification and maintenance)
- Treatment costs (including modifications)
- Sludge handling and disposal
- Resource recovery opportunities

Of the above, population density and the refuse generation rate were considered of major importance. Population density (i.e., single-family or multi-family dwellings) affects the technological choice for a neighborhood versus an individual household grinder system. The latter appeared feasible in low density areas, whereas neighborhood grinder systems are more practical for density areas of four or more households per structure.

Regarding refuse generation rates, the cost comparisons between conventional and wet transport systems in Table 2 are based on 1.02 kg (2.25 pounds) per capita per day; EPA estimated a national average generation rate of 1.51 kg (3.32 pounds) for 1972.[7] As shown in Table 1, this assumption has some effect on total system cost. These rates do not include commercial and industrial solid wastes.

The largest single cost component of the wet transport system is the annualized grinder and associated system costs of $80 to $105 (Table 2). The initial expense for the grinder and the associated installation cost is crucial to consumer acceptance of the entire concept. An underlying assumption is, therefore, that all homeowners would voluntarily purchase the grinder and assume the necessary installation costs. Otherwise, adoption of the system would have to be mandatory to be workable. In new housing areas just being developed, the system could perhaps be made a part of the development plan. The extent of plumbing modifications may make it prohibitive for older, established residential areas.

The nongrindable component of household refuse (e.g., bulky items, metals, resilient plastics, rubber, leather, synthetic textiles, etc.) would have to be collected in the usual manner, but at less frequent intervals. Modifications in collection equipment might be needed since the nongrindables would contain a large proportion of metals and be of higher densities than conventional refuse. Storage of nongrindables would also have to be provided. If the trend toward increased use of paper and plastic continues, the proportion of metals in refuse would probably decline.

An important but not fully determined component is the necessary sewer maintenance cost. The cost analysis assumed only one sewer flushing each year, but it was noted that up to four flushings might be needed. This increase would quadruple the costs for sewer maintenance.

The cost and availability of water for the grinding system are additional variables to be considered. Initial flow must be diluted to below 2 percent solids for proper functioning until the ground refuse is delivered to the curbside sewer system where the normal sewage flow is probably sufficient for further transport. The use of household nonfecal wastewater was considered as an alternative source of water; however, the additional costs for plumbing modifications and storage appeared to make this alternative uneconomical.

Another important aspect concerns the treatment of the refuse-sewage mixture. Preliminary indications are that the refuse-sewage mixture can probably be handled by conventional treatment methods, with modifications to accommodate the additional solids loading. An unanswered question, however, is what inhibitory effects on the treatment process may result from the introduction of hazardous or toxic and nonbiodegradable substances by way of the refuse.

Disposal of refuse in sludge form was shown to be less costly than disposal by conventional landfilling because economies of scale result from combining refuse and sewage, according to the Curran results.[8] The larger volume of sludge from wet transport systems, however, increases the importance of selecting sites for sludge disposal. In the Springfield, Massachusetts, SMSA, refuse disposal, as combined sludge with conventional disposal of nongrindables, was estimated to cost $3.00 annually on a household basis, compared with $4.25 for refuse disposal by landfill and sewage sludge separately (Table 2). Incineration of combined sludge appeared to be economical when the distance to a landfill exceeded 72.4 kilometers (45 miles).

Resource recovery opportunities under a wet transport system (with the possible exception of paper) appeared to be equal to those existing under the conventional refuse handling system. From an energy conservation viewpoint, the increased use of electricity for grinding and treatment plant operations would be offset by a savings in fuel used by refuse collection vehicles. Additionally, the wet transport system may offer a better potential for energy recovery through methane production by anaerobic digestion.

CONCLUSION

Although the economics appear unfavorable at this time for the wet transport system, it has certain potential advantages over the conventional method of handling household refuse that are not easily converted into monetary terms. These include improvements in aesthetics, reductions in noise and odor pollution, and potential improvement in public health. Evaluating the true social desirability of the wet transport system is extremely difficult and requires the conceptualization and measurement of all costs and benefits. From society's point of view, the refuse management system to be preferred is that that maximizes total net social benefits, including the environmental impacts.

There are certain technological aspects about the wet transport system requiring further investigation. These include: (1) the degree of sewer maintenance required, (2) the compatibility of treatment systems with the heavier loading caused by refuse, including possible increases in hazardous and toxic substances, and (3) the extent to which modifications would have to be made to household plumbing and existing sewer lines.

Wide-scale implementation of the wet transport system is not likely to occur for some time. If it does occur, implementation will be gradual to minimize any adverse effects on refuse workers and provide additional time to evaluate the areas needing further study.

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WATER SUPPLY RESEARCH LABORATORY

May 23, 1975

THE BACTERIAL QUALITY OF BOTTLED WATER

Edwin E. Geldreich, Harry D. Nash, Donald J. Reasoner, and Raymond H. Taylor*

The growing public concern over water pollution, the fact that numerous municipal water supplies use polluted streams for source water, opposition to the addition of fluoride to some municipal water supplies, and the taste and odor problems in some community water supplies have stimulated growth of the bottled-water industry in the United States. The recent publicity about the occurrence of organohalides in finished drinking waters has also had a positive effect on sales. In 1972, there were an estimated 700 water-bottling plants in the United States that accounted for at least $107 million in total sales.

Some of the larger bottled-water companies now reach large regional markets throughout the country in addition to expanded local sales. With the increased geographical distribution and growth in bottled-water sales, an increasing proportion of the population is exposed to a potable water source of uncertain bacteriological quality. The need for systematic bottled-water quality surveillance is a growing concern. Some states require that bottled-water quality standards comply with the U.S. Public Health Service Drinking Water Standards;[1] however, most states either have no regulations, or the existing regulations are ineffective or not enforced. Unfortunately, few states specify a definite sampling frequency for bacteriological examination or repeat testing when the initial sample results are unsatisfactory.

The assumption that bottled water is of unquestionably good quality simply because the source water is a spring or artesian well is untenable. The labeling of some bottled waters may even imply that the bottled waters are derived from pristine sources that do not require treatment and consequently are superior to municipal water supplies that utilize polluted raw water sources.

The purpose of this study was three-fold:

(1) investigate the bacteriological quality of a variety of brands of bottled water purchased from retail outlets,
(2) investigate the variability in bacteriological quality of freshly bottled water, and
(3) characterize changes in the bacterial density of bottled water during storage.

CONDUCTING THE SURVEY

Sample Collection

One-half- or one-gallon containers of available brands of bottled water were purchased from local retail outlets for bacteriological examination. After initial examination, some bottled-water samples were examined repeatedly over a period of 30 days or longer to evaluate changes in bacterial density during storage.

In addition, samples of freshly bottled water, obtained directly from the bottler, were examined bacteriologically. A portion of these samples had been collected in conjunction with a nationwide bottled-water study conducted by the Water Supply Division, U.S. Environmental Protection Agency,[2] in which 25 water bottlers were surveyed. Five water bottlers in the Cincinnati, Ohio, area were included in that survey.

All bottled-water samples were transported (unrefrigerated) to the laboratory within 6 hr for analysis. Some samples of freshly bottled water were held in the laboratory at 23 ± 2 C for repeat sampling to determine how the bacterial populations changed during storage. After the initial examination, repeat examinations were conducted over a period of 63 days.

Bacteriological Tests

1. Standard Plate Count (SPC). The SPC of bottled-water samples (freshly bottled and retail purchased) was determined using the pour plate procedure described in *Standard Methods for the Examination of Water and Wastewater*.[3] Five replicate plates were made for each dilution. The plates were examined and colonies counted after 72-hr incubation at 35 C.

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2. Total and Fecal Coliform. Total and fecal coliform determinations were conducted according to *Standard Methods* membrane filtration procedures. Separate 250-ml volumes of samples were examined for fecal and total coliform determinations of bottled water.

3. *Pseudomonas aeruginosa* counts. The membrane filter method of Levin and Cabelli,[4] utilizing M-PA agar, was used to examine the bottled-water samples for the presence of *Pseudomonas aeruginosa*.

SURVEY RESULTS

A total of 129 freshly bottled-water samples were collected directly from 25 different bottlers. Of these only 14 (10%) had an initial SPC greater than 500 bacteria per ml (Table 1). Total coliforms were detected in six samples; only two of these exceeded the USPHS Drinking Water Standards coliform limit (for an individual sample) of four coliforms per 100 ml. Also, one of the two coliform positive samples contained fecal coliform bacteria, and the other coliform positive sample contained 52 *Pseudomonas aeruginosa* per 100 ml.

TABLE 1. RANGES OF STANDARD BACTERIAL PLATE COUNTS IN FRESHLY BOTTLED WATER SAMPLES*

SPC/ml	Number of samples	% of samples	Cumulative %
<1	23	17.8	17.8
1-10	56	43.4	61.2
10-100	28	21.7	82.9
100-500	8	6.2	89.1
500-1,000	5	3.9	93.0
1,000-10,000	7	5.4	98.4
>10,000	2	1.6	100.0

*All samples were plated within 48 hr of bottling.

The bacterial densities in bottled-water samples of unknown age purchased from retail sales outlets were highly variable. The data in Table 2 show that 42 of 101 samples (41.6%) contained greater than 500 bacteria per ml, and about 11% contained less than 1 bacterium per ml. Three samples had total coliform counts that exceeded the USPHS Drinking Water Standards. Fecal coliform bacteria were not detected, but one sample contained *Pseudomonas aeruginosa* at a concentration of 13 per 100 ml.

Table 3 shows the variability in bacterial density and the frequency of coliform-positive samples among the 19 brands of bottled water analyzed during this study. Examination of the bacterial density ranges reveals the variability of the SPC from brand to brand, as well as the variability within the same brand where multiple samples were analyzed.

TABLE 2. STANDARD BACTERIAL PLATE COUNTS IN BOTTLED WATER OBTAINED FROM RETAIL OUTLETS*

SPC/ml	Number of samples	% of samples	Cumulative %
<1	11	10.9	10.9
1-10	20	19.8	30.7
10-100	19	18.8	49.5
100-500	9	8.9	58.4
500-1,000	6	5.9	64.3
1,000-10,000	16	15.9	80.2
>10,000	20	19.8	100.0

*Samples purchased after undetermined periods in stock.

Comparison of the data in Tables 1, 2, and 3 shows that the bacteriological quality of bottled water examined within 48 hr of bottling was not representative of the quality of bottled water obtained from the retail shelf after an undetermined period in stock. Forty-four bottled-water samples purchased from retail stores were stored at room temperature for 30 days and sampled weekly. The maximum SPC per ml recorded for stored bottled-water samples are grouped by range in Table 4. The SPC in 33 of the 44 stored samples (75%) increased to over 500 bacteria per ml during the storage period; this indicates that sufficient nutrients were present in the bottled waters to support significant increases in bacterial densities.

In addition, changes in bacterial populations of six freshly bottled-water samples stored at 23± 2C were followed for a period of 63 days. These samples were collected and examined in conjunction with a nationwide pilot survey of water bottlers and bottled water.[5]

To determine if the increase in bacterial density in bottled water could be minimized during storage, seven pairs of bottled-water samples were purchased. One bottle from each pair was stored at refrigerator temperature (4±2C) and the other stored at room temperature (23± 2C) for the duration of the storage experiment. Samples from each bottle were examined weekly for 8 wk to observe changes in bacterial density. Results from six of the paired samples showed that the maximum bacterial density attained in the stored refrigerated sample was significantly lower than that attained in the stored unrefrigerated sample (Figure 1). In the remaining paired sample, maximum counts of less than 20 bacteria per

TABLE 3. COMPARISON OF STANDARD PLATE COUNT VARIABILITY AMONG BRANDS OF BOTTLED WATER

Brand	Sample type*	Number of samples	SPC/ml range†	Coliform positive samples‡
A	F	91	<10-25,000	3/91
	R	16	<10-28,000	0/16
B	R	6	<10-260,000	0/4
C	F	1	<10	0/1
	R	11	1200-160,000	0/11
D	R	5	31-650	0/5
E	F	1	<10	0/1
	R	2	100,000-390,000	2/2
F	R	2	<10-12,000	0/2
G	F	2	<10	0/2
	R	1	<10	0/1
H	R	9	<10-390	1/9
I	R	2	<10	0/2
J	R	2	<10-2,000	1/2
K	R	2	<10-12	1/2
L	R	4	13-4300	0/4
M	R	1	<10	0/1
N	R	1	<10	0/1
O	R	1	1,000,000	1/1
P	R	1	8,600	0/1
Q	R	1	<10	0/1
R§	R	1	<10	0/1
S§	R	1	12	0/1

*Fresh (F) = Samples direct from bottler, examined within 24 hr; Retail (R) = Samples of unknown age purchased from retail outlets.

†Standard Plate Count (SPC) range values represent average bacterial counts per ml, calculated from five replicate plates, incubated for 72 hr at 35 C, using plate count agar.

‡One or more coliforms/100 ml.

§Imported bottled water, carbonated.

TABLE 4. MAXIMUM STANDARD PLATE COUNTS ATTAINED IN RETAIL PURCHASED BOTTLED WATERS DURING ROOM TEMPERATURE STORAGE FOR 30 DAYS

Maximum SPC/ml	Number of samples	% of samples	Cumulative %
<1	2	4.5	4.5
1-10	4	9.1	13.6
10-100	4	9.1	22.7
100-500	1	2.3	25.0
500-1,000	1	2.3	27.3
1,000-10,000	13	29.5	56.8
>10,000	13	43.2	100.0

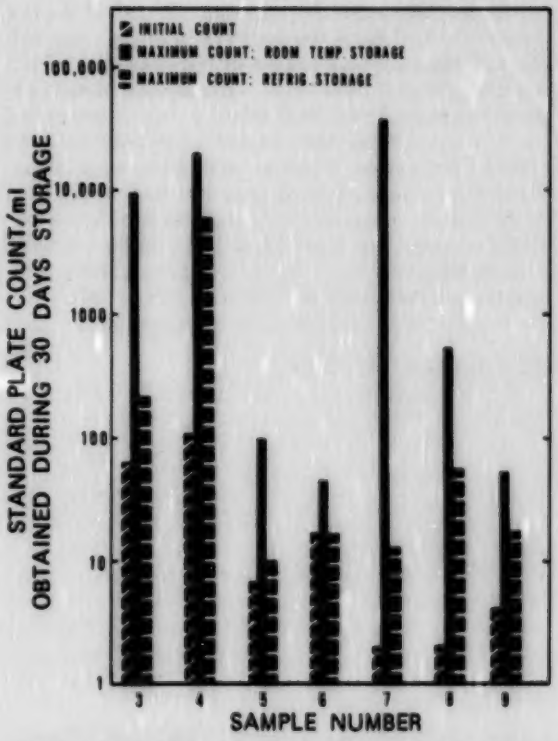

Figure 1. The effect of refrigerated and room temperature storage on the standard plate count of bottled water.

ml were observed in both the room temperature and refrigerated samples. The peak bacterial density in the refrigerated samples occurred after an average storage of 26 days, as opposed to only 11 days storage for the room temperature samples.

DISCUSSION

Although a specific statement may not be present on the brand label of a bottled-water product, it may be implied that, compared with municipal supply source water, the bottled-water source is initially of better quality and the finished product will retain this high quality during shelf life prior to purchase. The results obtained during this study do not support this implication. Persistence of initial bacterial contamination will be determined by the availability of bacterial nutrients, water temperature, pH, and antagonism or competition of other bacteria present in the water.

Concern over high bacterial counts in bottled water relates to: the possible loss of coliform test sensitivity in waters with excessively high bacterial populations and the increased risk of human exposure to organisms that are considered secondary pathogenic invaders.[5] The bottled-water standards adopted by the U.S. Food and Drug Administration,

which became effective on May 22, 1974,[6] do not include a SPC limit for bacteria, although the coliform limit is similar to that of the present USPHS Drinking Water Standards. This is unfortunate because, in many cases, bottled water is recommended for use in preparing baby formula, prescriptions, and fruit juices, coffee, tea, and other beverages. Thus, some infants and many older people who use bottled water are in a position of increased health risk because of possible exposure to bacteria that may be secondary pathogens. Both young and elderly individuals are relatively more susceptible to secondary pathogens than the rest of the population.

RECOMMENDATIONS

Bottled drinking water should be analyzed for bacteriological quality at the same frequency per month as is required by the USPHS Drinking Water Standards, with a repeat sampling program and a followup sanitary survey when the results are unsatisfactory. At the time of bottling, the water should contain less than 1 coliform per 100 ml and have a SPC of less than 500 bacteria per ml. Samples taken from supermarket, drugstore, and restaurant supplies should have less than 1 coliform per 100 ml and a SPC of less than 1,000 organisms per 1 ml. Establishment of a SPC limit is suggested for control of bacterial quality deterioration during storage.

The results from the bottled-water survey and from the storage studies support two recommendations for the bottled-water industry and retailers. First, refrigerated storage will minimize bacterial multiplication in bottled water from the time of bottling to sale. The consumer should also be advised to keep bottled water refrigerated. Second, each container should be marked with the bottling date, or lot number, or both to assist both retailer and consumer. The lot number would be desirable as an aid to recall of a given lot of bottled water from retail shelves when the bacteriological quality is unacceptable and, at the same time, would discourage retail overstocking, shorten storage time, and thus, minimize increases in bacterial populations in bottled-water products.

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MODERN WAYS OF STRIP MINING IN MOUNTAINOUS AREAS

Elmore C. Grim*

BACKGROUND

The energy crisis has placed a great demand on the coal resources of America. A U.S. Bureau of Mines report estimates that the demand for coal could grow from the 595 million tons produced in 1973 to 1.5 billion tons a year by 1985. Half of the industry's production now comes from surface mines. To achieve the 1985 figure, surface mining will expand, since it takes 5 years to open a new deep mine and only 2 years to open a surface mine.

Surface mining is a very broad term that refers to any process of removing the earth, rock, and other strata to uncover the underlying mineral or fuel deposit. Strip mining is a type of surface mining in which the strata above the deposit are removed in narrow bands, one cut at a time. By its very nature, strip mining is a destructive process.

Strip mining in the past has created problems that are still present today. Mining practices have all too often been aimed at removing minerals by the simplest and cheapest method possible, without plans for the preservation of land, water, and air, and with too little consideration for the rights of others. Recent laws have resulted in new attitudes and methods by the mining industry and improved environmental conditions.

Strip mining methods employed to recover coal can be divided into two general types: area and contour.

Area strip mining is practiced on gently rolling to relatively flat terrain and is commonly found in the Midwest and Far West. It is characterized by giant earthmoving equipment capable of handling several thousand cubic yards of material per hour. Unless graded or leveled, the mined area resembles a plowed field or the ridges of a gigantic washboard (Figure 1). In general, the pollution from area mines is not as severe as that from contour mines. Silt from erosion can often be confined to the mining area. The current legislative trend is to require restoration of the disturbed area to its approximate original contour with all spoil ridges and highwalls eliminated and no depressions left to accumulate water. Several States now require the operator to separate topsoil from the subsoil and to stockpile the two types separately. When mining is completed, the overburden can then be put back in its original sequence and revegetated to prevent erosion.

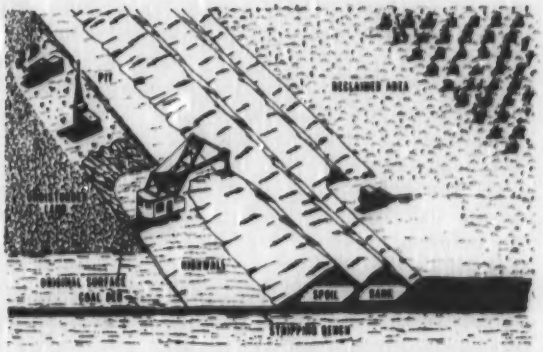

Figure 1. Area strip mining with concurrent reclamation.

Contour strip mining is practiced on rolling to very steep terrain. The conventional method of mining consists of removing the overburden from above the mineral seam, starting at the outcrop and proceeding around the hillside (Figure 2). The cut appears as a contour line, hence the name. A shelf or bench is created on the side of the hill. On the inside, it is bordered by the highwall, ranging in height from a few feet to more than 100 feet; on the outer side, the bench is bordered by a high ridge of spoil with a precipitous downslope that is subject to severe erosion and landslides. Another problem inherent in this type of mining is the toxic materials in the overburden. During the stripping operation, the high-quality soils near the surface are placed on the bottom of the spoil pile and then covered with low-quality and

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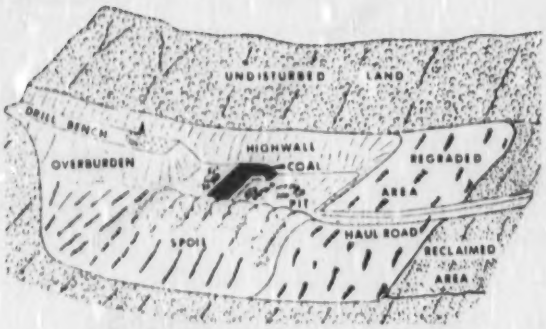


Figure 2. Typical contour strip mining.

often toxic overburden materials. These materials are exposed to weathering and conversion to soluble acids that are carried away by water. Erosion prolongs the pollution problem by continuing to reveal new surfaces to weathering.

Legislation at both the State and Federal level is becoming more stringent and making it illegal to push overburden beyond the outcrop and over the mountainside. This type of restriction will ban the conventional method of strip mining on steep slopes as well as other mining methods that create a fill section beyond the solid edge of the bench. Because of this, concerned Federal and State agencies, along with the coal industry, have been working together to develop modern contour mining methods that minimize the adverse effects on the environment and at the same time allow the maximum recovery of coal. These new methods (head-of-hollow fill, mountain-top removal, and block cut) are now accepted methods of mining on steep slopes. The spoil is not placed on the outslope, but hauled to a fill area designed for that purpose or placed on the bench behind the operation. These methods are not the final answers for all mining conditions and are being refined as more experience with varying conditions is gained.

MODERN CONTOUR MINING METHODS

Head-of-Hollow Fill Method

The head-of-hollow fill method was developed to provide storage space for spoil from the removal of entire mountain tops and as a waste area for overburden from contour benches. Instead of miles of unstable outslope, with its potential for slides and erosion, a stable usable area can be constructed. This method, if used properly, will also improve aesthetics, allow for full recovery of one or more coal seams, and produce potentially valuable flat-to-rolling mountaintop land.

Narrow V-shaped, steep-sided hollows that are near the ridge top and are free of underground mine openings, seeps, or wet weather springs are selected for filling. The size of the selected hollow must be such that the spoil generated by the mining operation will completely fill the treated head-of-hollow (Figure 3).

If the head-of-hollow is constructed according to design, stability of the fill can be expected. The horizontal and vertical pressures should provide adequate friction to prevent a failure in the fill. Several head-of-hollow fills have passed through five

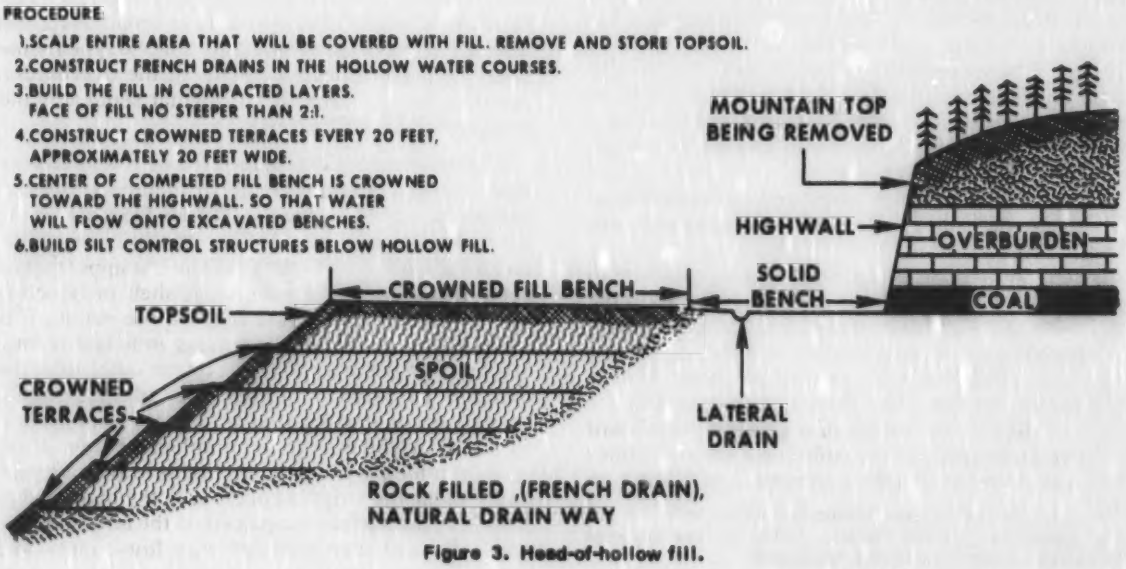

PROCEDURE
1.SCALP ENTIRE AREA THAT WILL BE COVERED WITH FILL. REMOVE AND STORE TOPSOIL.
2.CONSTRUCT FRENCH DRAINS IN THE HOLLOW WATER COURSES.
3.BUILD THE FILL IN COMPACTED LAYERS. FACE OF FILL NO STEEPER THAN 2:1.
4.CONSTRUCT CROWNED TERRACES EVERY 20 FEET, APPROXIMATELY 20 FEET WIDE.
5.CENTER OF COMPLETED FILL BENCH IS CROWNED TOWARD THE HIGHWALL. SO THAT WATER WILL FLOW ONTO EXCAVATED BENCHES.
6.BUILD SILT CONTROL STRUCTURES BELOW HOLLOW FILL.
MOUNTAIN TOP BEING REMOVED
HIGHWALL
OVERBURDEN
SOLID BENCH
COAL
CROWNED FILL BENCH
TOPSOIL
SPOIL
CROWNED TERRACES
LATERAL DRAIN
ROCK FILLED ,(FRENCH DRAIN), NATURAL DRAIN WAY

Figure 3. Head-of-hollow fill.

winters with no slides and little or no erosion. Some operators have graded the face of the fill to approximately 22° from the horizontal, eliminating the crowned terraces.

Mountain-Top Removal Method

The mountain-top removal method of surface mining is an adaptation of area mining to contour mining for rolling-to-steep terrain. Where coal seams lie near tops of mountains, ridges, knobs, or knolls, they can usually be economically strip mined. The entire tops are removed down to the coal seam in a series of parallel cuts. Excess overburden that cannot be retained on the mined area is transported to head-of-hollow fills, stored on ridges, or placed in natural depressions. This mining method produces large plateaus of level, rolling land that may have great value in mountainous regions.

The first cut is stripped as a box cut, leaving at least a 15-foot barrier of coal bloom undisturbed (Figure 4). This cut is made roughly parallel to the ridge. Overburden from the first cut is transported to the predetermined storage area. However, the overburden from succeeding cuts is deposited in the previously excavated cut. The mountain top is thus reduced by a series of cuts parallel to the ridge line. Approximately 50 percent of the overburden is transported to storage areas for disposal, and none is pushed over the downslope.

Figure 4. Mountain-top removal method: first cut (box cut).

When mining is completed, the mountain top is completely covered with a 20- to 40-foot layer of spoil and is graded nearly flat (Figure 5). Coal is recovered from areas that would not ordinarily be mined because they are unsuitable for underground mining. Since all the coal is recovered, the reclaimed area will not be disturbed again by future mining.

Block-Cut Method

The block-cut method is a simple innovation of the conventional contour strip mining method for steep terrain. Instead of casting the overburden from above the coal seam down the hillside, it is hauled back and placed in the pit of the previous cut. The method is not new and is known by various names (haul back, pit storage, put and take, etc.), depending on the locality. Basically, the operational proce-

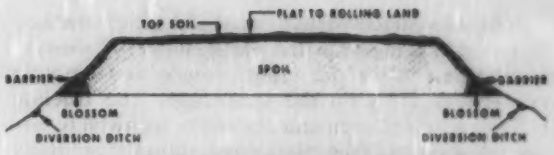
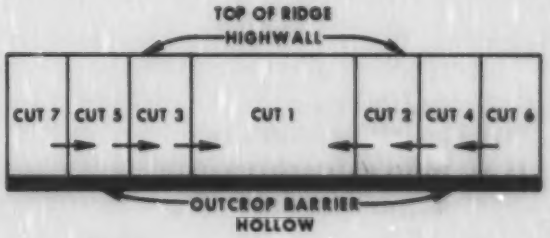


Figure 5. Mountain-top removal method: mountain top after final grading and topsoiling.

dures are similar in that no spoil is deposited on the downslope below the coal seam, topsoil is saved, overburden is removed in blocks and deposited in prior cuts, the outcrop barrier is left intact, and reclamation is integrated with mining.

When beginning the mine, a block of overburden is removed down to the coal seam and disposed of. This first-cut spoil can be placed above the highwall or moved laterally and deposited in a head-of-hollow fill or ridge fill. The original cut is made into the hillside to the maximum depth that is to be mined. The width is generally three times that of the following cuts. After the coal is removed, the overburden from the second cut is placed in the first pit, and the coal from the second cut is removed. This process is repeated as mining progresses around the mountain. Once the original cut has been made, mining can be continuous, working in both directions around the hill or in only one direction (Figure 6).

The cuts are mined as units, thereby making it easier to retain the original slope and shape of the mountain after mining. In all cuts, an unmined outcrop barrier is left to serve as a notch to support the toe of the backfilled overburden. Block-cut mining makes it possible to mine slopes steeper than those presently mined without the danger of slides and with minimal disturbance. Approximately 60 percent less total acreage is disturbed by this method than by other mining techniques now in use. There is



PROCEDURE:
1. SCALP FROM TOP OF HIGHWALL TO OUTCROP BARRIER. REMOVE AND STORE TOPSOIL.
2. REMOVE AND DISPOSE OF OVERBURDEN FROM CUT 1.
3. PICK UP COAL, LEAVING AT LEAST A 15 FOOT UNDISTURBED OUTCROP BARRIER.
4. MAKE SUCCESSIVE CUTS AS NUMBERED.
5. OVERBURDEN IS MOVED IN THE DIRECTION, AS SHOWN BY ARROWS, AND PLACED IN THE ADJACENT PIT.
6. COMPLETE BACKFILL AND GRADING TO THE APPROXIMATE ORIGINAL CONTOUR.

Figure 6. Block-cut method.

significant visual evidence that the block-cut method is less damaging than the old practice of shoving overburden down the mountainside and creating permanent scars on the landscape. The treeline below the mined area and above the highwall is preserved. Results of the mining operation are generally hidden and cannot be seen from the valley below. This cosmetic feature is only one of the advantages that make this an acceptable steep-slope mining method.

The block-cut method is no longer experimental and is now operational in several States. Enough information is available from active operations to show that this method is potentially feasible from an economic and environmental standpoint.

CURRENT RESEARCH

A comprehensive environmental assessment of the head-of-hollow fill, mountain-top removal, and block-cut methods has never been made. Such an assessment is necessary to develop a plan of action to help control pollution from surface mining activities and to determine land use capabilities. The Mining Pollution Control Branch is currently making these assessments. Guidelines developed as a result of this program will provide a means for selecting methods, techniques, and alternatives for premining planning, operating procedures, and reclamation. Controls for eliminating significant problems documented during the assessment will be recommended.

With proper surface mining and reclamation practices, the Nation can have the fuel it so vitally needs and minimize environmental degradation.

PERTINENT PUBLICATION

The Mining Pollution Control Branch has recently published a comprehensive manual on the strip mining of coal:

Environmental Protection in Surface Mining of Coal. by Elmore C. Grim and Ronald D. Hill (Environmental Protection Technology Series, EPA-670/2-74-093, October 1974). Copies are available from the Technical Information Staff or the Mining Pollution Control Branch, National Environmental Research Center, Cincinnati, Ohio 45268; or from the National Technical Information Center, Springfield, Va. 22151 as report PB 238 538/AS.

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SOLID AND HAZARDOUS WASTE RESEARCH LABORATORY

June 30, 1975

NATURE AND USE OF COAL ASH FROM UTILITIES

Richard A. Carnes*

BACKGROUND

In 1972, an estimated 46 million tons (or more) of coal ash were collected from the burning of 350 million tons of coal in some 500 power plants in the United States. The ash is derived from the inorganic mineral constituents in the coal and the incompletely combusted organic material. Bottom ash is collected from the bottom of the boiler unit and fly ash is taken from the air pollution control equipment through which the stack gases pass.

The largest concentration of coal-burning power plants in the country is in the Middle Atlantic States and the East North Central States; west of the Mississippi, coal-burning plants constitute only a small percentage of the total. A geographic distribution of coal burned by U.S. utilities in 1971 is shown in Table 1.

TABLE 1. GEOGRAPHIC DISTRIBUTION OF COAL BURNED BY U.S. UTILITIES IN 1971

Geographic area	Coal burned (thousands of tons)	Percent of total
New England[a]	2,701	0.8
Middle Atlantic[b]	43,869	13.4
East North Central[c]	119,444	36.5
West North Central[d]	27,898	8.5
South Atlantic[e]	65,903	20.1
East South Central[f]	51,565	15.7
West South Central[g]	10	0.1
Mountain[h]	15,885	4.9
Pacific[i]	0	0
Total	327,358	100.0

[a]Maine, New Hampshire, Vermont, Massachusetts, Rhode Island, Connecticut.
[b]New York, New Jersey, Pennsylvania.
[c]Ohio, Indiana, Illinois, Michigan, Wisconsin.
[d]Minnesota, Iowa, Missouri, Kansas, Nebraska, South Dakota, North Dakota.
[e]Maryland, Delaware, Virginia, West Virginia, North Carolina, South Carolina, Georgia, Florida.
[f]Mississippi, Alabama, Tennessee, Kentucky.
[g]Louisiana, Arkansas, Texas, Oklahoma.
[h]New Mexico, Arizona, Nevada, Colorado, Utah, Wyoming, Montana, Idaho.
[i]California, Oregon, Washington.

CHARACTERISTICS OF COAL ASH

Chemical Constituents

The specific chemical composition of a particular coal ash is dictated by the geology of the coal deposit and the boiler unit operating conditions. The coal ash residues recovered from various boiler units are primarily iron aluminum silicates with additional amounts of lime (CaO), magnesia, sulfur trioxide, sodium oxide, and carbon (see Table 2). About 12 percent of the coal burned is recoverable as coal ash residue. A high percentage (50-90) of that ash is in the glass state, with small quantities of quartz, mullite, magnetite, and hematite mineral phases.

Physical Characteristics

About 70 percent of the coal ash residue is collected as fly ash. For any specific boiler unit, the fly ash and bottom ash have basically the same chemical composition except that bottom ash has a reduced carbon content. Fly ash generally occurs as fine spherical particulates with an average particle diameter of 7 microns. Depending on carbon content, fly ash ranges in color from light tan to black and has an average specific gravity of 2.3. The pH of fly ash varies from 6.5 to 11.5, but averages about 11. Almost 20 percent of fly ash, by volume, is composed of very lightweight particles that float on the surface of an ash lagoon. These particles, called cenospheres, have a true density of about 0.5 g per

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TABLE 2. CHEMICAL CONSTITUENTS OF COAL ASH

Constituent	Average %	Constituent	Average %
Silica (SiO_2)	45	Boron (B)	Trace
Alumina (Al_2O_3)	25	Phosphorus (P)	Trace
Ferric oxide (Fe_2O_3)	14	Manganese (Mn)	Trace
Calcium oxide (CaO)	4	Molybdenum (Mo)	Trace
Magnesium oxide (MgO)	2	Zinc (Zn)	Trace
Titanium dioxide (TiO_2)	1	Copper (Cu)	Trace
Potassium oxide* (K_2O)	2	Mercury (Hg)	Trace
Sodium oxide (Na_2O)	1	Uranium (U)	Trace
Sulfur trioxide (SO_3)	2	Thorium (Th)	Trace
Carbon (C)	4		

*Alkalies

cm^3. These cenospheres are CO_2- and N_2-filled microspheres of silicate glass ranging in size from 20 to 200 microns.

The bottom ash is collected either as an ash or a slag, depending on the particular boiler design. The ash material is gray to black in color, is quite angular, and has a porous surface. The slag particles are normally black, angular particles having a glassy appearance. The bottom ash particles have an average diameter of 2½ mm and an average specific gravity of 2.5.

Coal Ash Characteristics and Advanced Technology

Advancing boiler-design technology and the enforcement of stricter air pollution codes for boiler facilities may alter the nature of the coal ash produced in the future. The various desulfurization processes, coal fractionation processes, and new designs for electric generating facilities can produce coal ash and slag products that are considerably different from those currently being produced.

The coal fractionation processes used for obtaining clean gas or liquid fuels and the reconstitution of the coal to obtain a clean, low-ash, low-sulfur fuel yield slag and char residues at the conversion facility rather than at the power plant. The liquefaction process produces a filter cake of inorganic materials. The fluidized-bed gasification process generates a powder waste composed of the fluidizing media, the coal residue, and a calcium sulfate precipitate. A glassy slag is produced from the high-temperature gasification process. Because of the relative newness of these processes, the chemical composition and physical characteristics of the residues have not yet been well defined.

Conversion of existing boiler units to fluidized-bed units will change the nature of the coal ash recovered. Ash from this process will be less vitrified, basically as a result of lower operating temperatures. The quantity of crystalline material will show an increase (quartz, magnetite, alumina, and calcium sulfate), and the alkaline content is also likely to be higher.

Several processes have been developed to meet the newly established codes for control of SO_2 emission from stationary sources. A number of these processes completely alter the nature of the collected material, and others add a new residue material to the solid waste stream. Most processes require the wet or dry injection of an alkaline powder (limestone, dolomites, etc.) to absorb the gaseous sulfur in the stack effluent. The presently preferred system is the wet injection or scrubbing process (limestone), which in most cases generates a new waste product ($CaSO_4$) rather than modifies the fly ash. Preliminary calculations indicate that these wastes will probably result in doubling the residue wastes from utilities. Since these desulfurization processes are still largely in the development or pilot stage, it is not possible to accurately identify the chemical characteristics at this time.

UTILIZATION

Since 1966, the coal ash utilization rate has varied only slightly and is estimated to be about 15 to 16 percent of the total ash collected in the United States. From data supplied by the Edison Electric Institute, it is apparent that the largest use of coal ash is as a mineral fill material for roads and other construction projects. Average European utilization of bituminous coal ash from 1972 was approximately 27 percent; but in Belgium, France, Poland, the United Kingdom, and West Germany, total utilization exceeded 50 percent. The two largest applications of European coal ash were as fillers on construction sites and in concrete blocks.

Though a multitude of technically sound applications have been developed for using coal ash, its total use has been very limited. Yearly fluctuations in the quantities of coal ash used in the various applications developed suggest that firm markets have not or cannot be established. At the present time, appreciable quantities of coal ash are being used only for fill for roads and other sundry construction projects. The use of coal ash as a replacement for cement in concrete and concrete products is making some progress, and a more stable market is being established. The use of fly ash in concrete offers several technical advantages—for example, improved mechanical strength and improved resistance to sulfate leaching. Fly ash and boiler slag are also being used to a large extent in road-base stabilization and as filler in asphalt. Boiler slag is particularly noted for increasing the skid resistance of asphalt pavement. The use of coal ash as a raw material in the

manufacture of Portland cement is another application that has recently increased. Recent research results indicate that large quantities of coal ash can also be effectively used for agriculture, land, and water reclamation projects. Fly ash has been effectively utilized in reclaiming surface mine spoil, as a soil nutrient, and as an aid in treating polluted waters.

A number of applications developed for coal ash have the potential for utilizing the entire quantity of ash generated. These include agriculture and land recovery, road-base stabilization, structural fill, and cement and concrete products.

Effective utilization of coal ash in the many defined applications requires that the potential user be favorably impressed with the product and that the product be economically advantageous. Depletion allowances, discriminatory freight rates, and other practices presently mediate against the economic competitiveness of coal ash. New or improved Federal economic policies toward secondary materials such as coal ash could substantially increase their use potential.

CONCLUSIONS AND RECOMMENDATIONS

Of the approximately 46 million tons of coal ash collected in 1972, only about 16 percent was utilized. Thus, over 38 million tons of ash had to be removed to disposal sites at the expense of the utility and, eventually, the consuming public. At present, disposal costs are approaching $2 per ton of ash disposed. To meet anticipated energy requirements for 1980, coal consumption by the utilities is projected to be almost 500 million tons. Such an increase in coal consumption coupled with the decreasing quality of available coal (higher ash content) will result in substantially increased quantities of coal ash. Stricter air pollution codes (reduced particulate and sulfur emissions) will also result in increased quantities of coal ash to be collected for ultimate disposal.

The technology for various uses of coal ash has been well established. The potential market for most of these applications is quite good, and several of these applications have potential markets that could utilize all the ash collected. The major need at this time is to initiate programs that will encourage greater use of the developed applications. With the anticipated increase in coal ash production and increasing disposal costs, the need for programs to stimulate ash utilization is paramount.

In addition to these use-stimulation programs, studies are needed to characterize ash residue from the new processes such as fluidized-bed boiler units, gasification and liquefaction, and desulfurization. Without such studies, effective utilization technology for these types of waste will not become available, and they will add significantly to the disposal problem.

A more complete discussion of the material presented here and an extensive bibliography are to be found in *Characterization and Utilization of Municipal and Utility Sludges and Ashes: Volume III Utility Coal Ash (EPA-670/2-75-033c)* prepared under the direction of N. L. Hecht and D. S. Duvall of the University of Dayton Research Institute for the Solid and Hazardous Waste Research Laboratory, NERC-Cincinnati, EPA Research Grant Number 800432, Richard A. Carnes, Project Officer.

USEPA OFFICE OF RESEARCH AND DEVELOPMENT—REORGANIZATION

Effective June 29, 1975, the new Office of Research & Development (ORD) organization became effective. This means that the original four National Environmental Research Centers and old laboratories were totally replaced by the new structure of 15 laboratories.

The new laboratories under the supervision of acting directors in Cincinnati carry out work either through its own facilities, field stations, or through contract, grant, or interagency agreements with other organizations.

ENVIRONMENTAL MONITORING AND SUPPORT LABORATORY (EMSL)
Mr. Dwight G. Ballinger

This laboratory develops field and laboratory methods for monitoring physical, chemical, radiological, microbiological, and biological water quality and waste characteristics. An Agency-wide quality assurance program is conducted by the laboratory, and technical support is provided to Regional and other laboratories on water and waste monitoring methodology and quality assurance techniques. The laboratory publishes methods manuals, quality assurance guidelines, and research reports on Agency water monitoring technology.

MUNICIPAL ENVIRONMENTAL RESEARCH LABORATORY (MERL)
Dr. Andrew W. Breidenbach

MERL is responsible for research, development, and demonstration programs for municipal wastewater, water supply, solid and hazardous wastes, and ancillary environmental problems. The programs include collection, treatment, disposal, and management of liquid and solid wastes, and the technology needed to produce potable water. The MERL effort includes an integrated systems and economic analysis approach to assure cost effective output for the municipal user. The Laboratory reports to the Deputy Assistant Administrator for Air, Land, and Water Use.

HEALTH EFFECTS RESEARCH LABORATORY (HERL)
Dr. R. John Garner

Another Cincinnati laboratory is designed to carry out research programs to determine the health effects of environmental pollutants and to assess the socioeconomic consequences of these effects. Toxicological, clinical, and epidemiological studies are conducted on a long- and short-run basis with support agencies assisting. The activities of the laboratory are supervised by a director who is responsible to the Deputy Assistant Administrator for Health and Ecological Effects.

INDUSTRIAL ENVIRONMENTAL RESEARCH LABORATORY (IERL)
Dr. David G. Stephan

Develops and demonstrates cost-effective technologies to prevent, control, or abate pollution from operations with environmental impacts associated with the extraction, processing, conversion, and utilization of energy and mineral resources and with industrial processing and manufacturing. Evaluates environmental control alternatives, including conservation measures, and assesses the environmental and socioeconomic impacts, of the operations. The Director of the laboratory is responsible to the Deputy Assistant Administrator for Energy, Minerals, and Industry.

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HEALTH EFFECTS RESEARCH LABORATORY

September 29, 1975

A RADIOTELEMETRY SYSTEM FOR MONITORING RESPIRATION IN DOGS

Mildred J. Wiester, Rumult Iltis, and Charles T. Pfetzer*

INTRODUCTION

The U.S. Environmental Protection Agency is interested in the health effects of many toxic pollutants in the atmosphere. Toxicological information can be obtained from animals chronically exposed to controlled experimental atmospheres containing these pollutants. Meaningful measurements of respiration in test animals are needed for this type of information. Since respiratory measurements are less altered if they can be obtained from free roaming animals, there is a need for suitable instrumentation and methodology by which breathing parameters can be remotely monitored and assessed. A system that accurately measures the exposure of an inhaled pollutant without hindering the animal's ventilatory mechanisms could be of utmost value in air pollution research. This study describes such a system developed for use with dogs.

If rate (frequency, f) and depth (tidal volume, V_T) of pulmonary ventilation are measured, the minute respiratory volume (V_E) of a dog can be calculated by: $V_E = V_T \times f$. This reduces the problem of measuring inhaled dosages of any known atmospheric pollutant to a matter of simple arithmetic. For example: $V_E \times$ atmospheric pollutant concentration $\times$ time = total dose. Known exposures of any pollutant can be correlated with physiological or behavioral effects as well as with pathological tissue levels of the pollutant.

The spirometer,[1,2] plethysmograph,[3] pneumotach,[4,5] and the pneumograph band[6] have all been used successfully to obtain accurate breathing rates and volumes in acute respiration studies. Each of these devices, however, is unsuitable for chronic exposure experiments for one or all of the following reasons: (1) physical restraints or anesthesia needed to apply or maintain a device may alter the dog's body position, breathing patterns, and mechanics; (2) connections and/or wiring of devices often results in reduced mobility; and (3) a dog may remove any device attached to it (e.g., pneumograph band).

Many methods of short-term evaluation call for using a restrained dog—a condition unacceptable in chronic studies.[4] The ideal chronic exposure requires conditions that allow natural breathing without contact with the experimenter.[7] Since barbiturates depress pulmonary ventilation,[8] chronic exposure requires the dog to be unanesthetized while measurements are being made. To be functional, the system would have to satisfy the following requirements: (1) respiratory measurements should be taken from unanesthetized and unrestrained dogs; (2) extra stimuli to respiration should be generated only by the pollutant; (3) animals should be able to tolerate the monitoring device with no ill effects; (4) the system should be reliable, stable, and easy to operate throughout the period of chronic exposure (approximately 6 weeks); (5) equipment and procedure should be economical and sufficiently easy to assemble so that the system could be used repeatedly on small groups of dogs; and (6) the system should be easy to calibrate. The purpose of this study was to develop and validate an implantable system that would accurately measure breathing rate and volume in dogs and be appropriate for monitoring respiration during chronic exposure to aerosol pollutants. An ideal monitoring system would be a small, implantable respiration sensor signaling a radiotelemetry transmitter. A device such as this eliminates the need for restraints or anesthesia during measurement and locates wires and instruments inside the dog.

Because suitable radiotelemetry equipment was not available commercially nor was there any method for respiratory measurements of this nature referenced in the literature, we undertook to develop such a system.

EQUIPMENT AND PROCEDURES

The system has two main implantable parts: the sensor and the transmitter. The sensor unit consists of a semiconductor strain-gauge, which, when surgically mounted on the dog's rib, detects and converts breathing movement to an output voltage signal. The

*Dr. Wiester and Rumult Iltis (513-684-7488) are with the U.S. Environmental Protection Agency's Health Effects Research Laboratory in Cincinnati, Ohio. C. T. Pfetzer submitted (1974) part of this work to Wright State University, Dayton, Ohio, in partial fulfillment of his M.S. requirements.

signal is fed into the transmitting unit and amplified. The transmitted signal is received by a remote FM receiver and subsequently recorded (Figure 1).

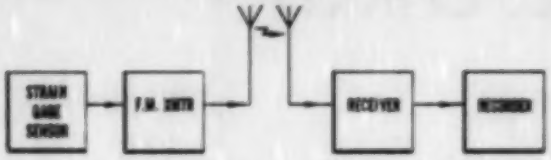
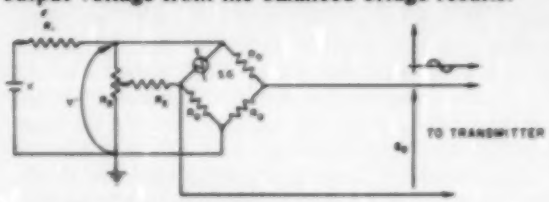


Figure 1. Sensor-radiotelemetry system.

The strain-gauge forms one arm of a Wheatstone bridge. The other arms are formed by three precision resistors equal in value to the nominal value of the unstressed gauge (Figure 2). When no stress is applied to the sensor, the output of the bridge is zero. Since the condition is difficult to maintain (because stresses are developed when the sensor is mounted on the rib), a balancing potentiometer trimmer is provided across the bridge. When stress is applied to the sensor, resistance of the gauge changes and an output voltage from the balanced bridge results.



R_1 = CURRENT LIMITING RESISTOR
R_B = BALANCING POTENTIOMETER (50K Ω)
R_O = NOMINAL RESISTANCE OF STRAIN GAGE (2·2K Ω)
SG = STRAIN GAGE
V = SUPPLY VOLTAGE (1·5 V)
GF = GAGE FACTOR (140)
$\mathcal{E}$ = STRAIN (IN/IN)
E_o = OUTPUT VOLTAGE

$$E_o = V \frac{\Delta R_O}{2 R_O} = V_s \, GF \, \mathcal{E} \text{ VOLTS}$$

Figure 2. Wheatstone bridge circuitry.

An ivory plate (30 × 6 × 2 mm) was chosen for the strain-gauge mounting. Gauges (BLH Electronics of Waltham, Mass.*) were the semiconductor type SPB 3-15-200 with a nominal resistance of 2,000 ohms and a gauge factor of 140. Eastman 910 adhesive was used for attaching the gauge to the ivory. Gauge leads were soldered to silver circuit stick tabs (Circuit Stick of Gardena, Calif.). The tabs were also attached to the plate with Eastman 910 adhesive. Belden 32 AWG lead wires were soldered to the standoff tabs. The surface of the plate containing the gauge and tabs were coated with a layer of Gaugekote (Bean and Associates, Detroit, Mich.), and the lead wires were covered with silastic medical grade tubing (Dow Corning). The unit and silastic tube connections were then encapsulated with Scotch cast #5 (3M Co.). This structure, when hardened, was hand sculptured to 34 × 10 × 4 mm with a hand rotary drill and cutting bits (Figure 3).

The small, single-channel FM transmitter (supplied by Case Western Reserve University, Cleveland, Ohio) was designed to respond to frequencies higher than those generated during normal breathing, which makes the transmitter nonresponsive to breathing signals. Therefore, an impedance-matching transformer was added between the strain-gauge and transmitter. Because the time constant of this circuitry is in milliseconds, this system is more than responsive to a 2- to 3-second breathing excursion. Also, AC coupling to the transmitter was required to eliminate effects of baseline shifts. The power source consisted of two Eveready (S76) silver oxide batteries. The batteries, Wheatstone bridge, impedance transformer, and transmitter were all built into a small, implantable unit 55 mm long and 9 mm in diameter.

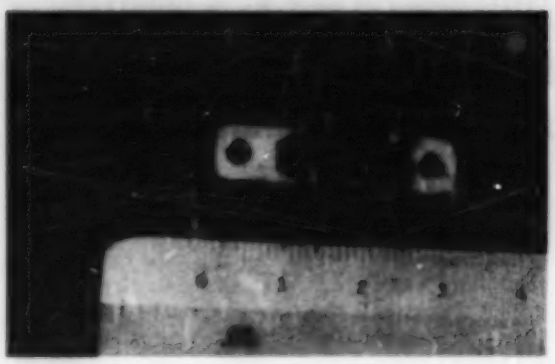


Figure 3. Mounted strain-gauge.

For reception of transmitted signals, a dipole-type antenna was designed and constructed. The antenna was insulated and mounted in the exposure chamber. A coaxial cable, connected to the antenna, extended to the FM receiver in a remote room. The receiver used was a commercially available (Defense Electronics) Model GPR 20 FM receiver. The AC signal decoded by its limiter discriminator is fed to a polygraph recorder.

Before surgery, all electronic implantable devices were gas sterilized. Surgery consisted of making a 3-inch (7.6 cm) incision in the lateral chest skin, retracting the muscles, and exposing a segment of the eighth rib. The periosteum was removed from the area of the bone where the sensor would be placed (just above and proximal to the anterior rib cartilage). With the use of orthopedic instruments, two holes were drilled through the rib, and the sensor was then secured with bone screws. During surgery, the resistance of the gauge was directly monitored by a digital ohmmeter. Screws were turned to full tightness in such a manner that the resistance of the gauge remained at its unstressed value and the sensitivity of the gauge was maintained at its maximum. Following strain-gauge attachment, wires from the gauge were threaded under the skin and exited through a puncture wound in the back neck skin. The transmitter unit was surgically implanted under the back skin; later it was attached to the sensor after the stability of the implanted strain-gauge apparatus was confirmed.

*Mention of commercial products does not imply endorsement by the U.S. Environmental Protection Agency.

SIGNAL INTERPRETATION AND COMMENTS

Calibration was aimed at achieving a good estimate of minute volume from the unanesthetized dog. To calibrate the strain-gauge signals for volume, the signals had to be compared with simultaneously recorded, direct-volume measurements, and for this purpose, a spirometer was used. Breathing output signals (seen in Figure 4B) have a time decay constant; therefore, they represent the rate of change of stress in the bone during breathing excursions. For calibration of breathing minute volume, the dog was anesthetized and outfitted with an endotracheal tube. The tube was placed in series with a Collins 13.6-liter spirometer (Collins Co., Braintree, Mass.). The dog was placed in three basic body positions: right lateral recumbency; left lateral recumbency; and stomach. For each position, the dog was monitored while breathing either room air or while breathing a low-oxygen, high-carbon-dioxide gas mixture, which caused increases in rates and volumes. The strain-gauge reading curves were analyzed for frequency and shape of the volume curve; they were also electronically integrated (Figure 4C) to determine areas under the curve. Area values were compared with minute volumes simultaneously recorded on the spirometer (Figure 4A).

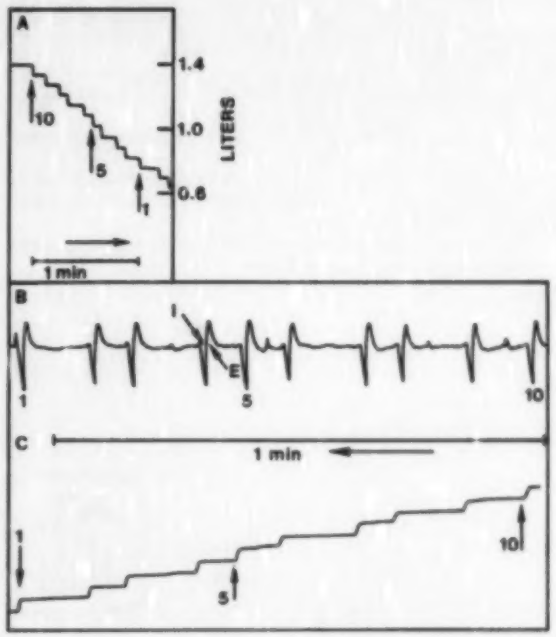


Figure 4. A, spirometer recording of 10 inhalation volumes; B, simultaneous breathing output; C, electronic integration of area under each inhalation curve.

A calibration curve was constructed from data obtained with the dog placed in different body positions and breathing different atmospheres. The curve plots minute volume (liters per minute), recorded by the spirometer against the area under the curve of a simultaneously recorded and integrated strain-gauge output (mm^2 per minute) (Figure 5).

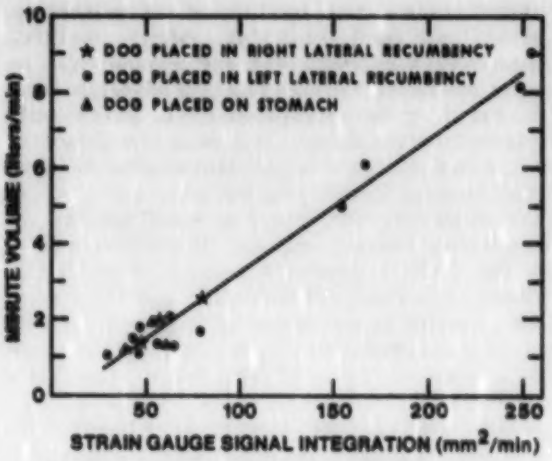


Figure 5. Strain-gauge volume calibration curve.

Results indicated that stresses developed in the ribs during breathing, detected by the strain-gauge, produced corresponding output voltages that showed a linear relationship to minute volumes. Furthermore, when the voltages were transmitted through the telemetry device, they retained their linearity.

Once calibration curves are constructed for each strain-gauge, the respective curves can be used to estimate volume measurements of subsequent strain-gauge signals. Practical application of a breathing rib sensor can be seen in Table 1.

TABLE 1. COMPARISON OF ANESTHETIZED AND UNANESTHETIZED BREATHING FROM THE SAME DOG.

Item	Dog respiratory values [a]	
	Before anesthesia	Immediately following pentobarbital anesthesia
Breathing rate (breaths/min)	16[b]	8.3[b]
Strain-gauge bridge output (mm^2/min)	38[b]	
Minute volume (liters/min)	1.01[c]	1.05[d]
Tidal volume (ml/breath)	63[c]	127[d]
Inhalation time (sec)	0.8[b]	0.8[b]
Exhalation time (sec)	1.6[b]	1.6[b]
Resting time (sec)	1.4[b]	5.4[b]

[a]Values were averaged over 3-minute periods.
[b]Read from polygraph recording.
[c]Computed using calibration curve (Figure 5).
[d]Directly read from Collins spirometer.

DISCUSSION

Both direct and indirect methods have been valuable in obtaining animal measurements of respiratory minute volume. If complete accuracy of results is paramount, it is best to use the direct methods. However, experimental design and procedure do not always allow for direct measurements, and in such cases, indirect methods are used at the expense of accuracy. In the case of chronic pollutant exposures, both direct and current indirect measurement procedures are unsuitable for many of the same reasons. Any methods presently available require regular

human contact, with handling of the animals and often, physical restraint. Many methods (pneumotach or spirometer, for example) also require anesthesia. Because of profound metabolic effects resulting from anesthesia, it is difficult to make inferences of physiological response to pollutants by the animal. For chronic pollutant studies, the trend must be away from human contact of any kind and toward an indirect system that would not alter the respiratory system in any way. It was the objective of this study to develop a monitoring device that allows for a close estimation of breathing volume with minimal experimenter contact. After analysis of the principles of breathing mechanics and a review of available systems, it was determined that a strain-gauge rib sensor should be developed.

The concept of a rib-stress sensor is based on the fact that the bony thorax has a bellows-like respiratory role. It alternately increases and decreases the volume of space that it encloses. The rib, a passive structure in the thorax, would generate stress only in response to the volume enclosed and would not reflect the numerous forces responsible for achieving this volume. In effect, the rib will respond to change of volume in the thorax regardless of whether the forces are due to intercostal or diaphragmatic sources.

Because of the circuitry of the system described earlier, the strain-gauge responding to stresses in the rib records only those dynamic changes of stress in the rib resulting from instantaneous changes during breathing. The strain-gauge then faithfully describes not only the exact time relationship and frequency of inspiration and expiration, but it also indicates the magnitude of volume change by the intensity of its output. The strain-gauge measures the rate of change in stress rather than stress duration in the rib, and subsequently the rate of change of volume in the thorax. Integration of this type of strain-gauge output yields respiratory volume measurements.

Other than signal interpretation and calibration of the sensor, a basic concern of the system is its durability after implantation. The sensor had to be hermetically sealed for protection against invasive body fluids. Various types of encapsulants and adhesives were considered, based on evaluation of over 40 different types by McKay.[9] Scotch cast #5 was chosen for its sealant properties, response to stress, availability, and general working convenience. The method of encapsulation used for this work prevented tissue fluid leakage for more than 8 weeks when properly applied.

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IMPROVING THE FUEL VALUE OF SEWAGE SLUDGE

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BACKGROUND

An increasingly crucial component of the cost of incinerating municipal sewage sludge is that contributed by consumption of auxiliary fuel.[1]

Raw liquid sludge, at approximately 4 percent solids, contains far too much water to permit autogenous combustion of the sludge solids. Even after mechanical dewatering in such devices as vacuum filters and centrifuges, mixtures of primary and secondary sludges still have moisture contents exceeding 75 percent. At these high moisture contents, more heat energy is required to evaporate the entrained water than can be provided by the sludge solids in the cake. Thus, auxiliary fuel, usually in the form of fuel oil or natural gas, is introduced into the incinerator to supply the necessary heat energy not provided by the sludge solids.

Varying auxiliary fuel requirements depend on the volatile content of the sludge and other constraints such as heat loss and noncombustible chemicals. Therefore, two variables are important when considering sludge incineration: (1) sludge volatility (combustibles) and (2) cake moisture content. Controlling either volatility or moisture, or both, is the key to improving the combustibility of the sludge cake.

Several municipalities are now exploring alternatives to costly consumption of expensive auxiliary fuels (fuel oil and natural gas). Among alternative fuels being considered are coal, municipal refuse, wood chips, and other paper products.

Biological, or secondary, sludge is much more difficult to dewater than is primary sludge. Most wastewater treatment plants that incinerate secondary and primary sludges blend the two together and dewater this mixture, usually on a vacuum filter. Because of the presence of secondary sludge, the mixture is very difficult to dewater. Cakes of low solids (14-20 percent) content are produced by vacuum filtration and relatively low filter rates result even though chemicals are normally used to aid dewatering.

A study[2] concerning the use of incinerator ash for sludge conditioning before vacuum filtration, completed in 1972, found that an ash dose of about 2 kg ash/kg dry sludge solids was required to double the yield of the unconditioned sludge. The moisture content (kg water/kg sludge solids) also increased with the higher dosages of ash. From a thermal standpoint, the ash, which has no beneficial heat value, represents a load of mass that was to be heated to temperature, thus absorbing energy needed for evaporating water. Therefore, auxiliary fuel would still be needed because the same amount of sludge solids and water are involved.

The Cedar Rapids (Iowa) wastewater treatment plant is now using incinerator ash along with chemicals (ferric chloride and lime) to improve the cake yield from a pressure filter. Addition of 1.5 kg ash/kg sludge solids has cut the chemical costs for conditioning from $22/metric ton[3] sludge solids to $7/metric ton sludge solids. However, at the ash ratios used, the volatile content of the cake is only 11 to 15 percent. Autothermic incineration is possible only because the high pressures in the filter press (200 psi) produce a cake with only 40 to 50 percent moisture.

Similar experience with ash conditioning and low polymer addition has occurred at the Indianapolis, Indiana,[4] wastewater treatment plant. Filter rates were increased almost three-fold and cake moisture dropped from approximately 80 to 60 percent. However, because of considerable buildup of inert ash, heavy additions of auxiliary fuel were still needed to complete combustion of the sludge cake.

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COAL AS CONDITIONER

Sludge dewatering aids, whether chemical, such as polyelectrolyte, or inert, such as ash, change either the chemical, electrical, or physical properties of the sludge before filtration. The most obvious effect of inert solids conditioning is a physical phenomenon. Sewage sludge (for the most part biological sludge) is a highly compressible substance. Under pressure, the floc particles tend to flatten out and expand. The result of this action is detrimental to the separation of water from the solids. Once the particles flatten out, the filter medium (the thin cake of flattened solids) becomes less porous and slows down the flow of filtrate; the calculated specific resistance of the cake increases under increasing pressure.

On the other hand, substances such as ash or granulated coal are, in essence, incompressible. The only appreciable resistance is that caused by a thicker cake deposited on the cloth as filtration proceeds. Therefore, to condition sludge with a substance such as ash or coal serves to reduce the original sludge compressibility.

THERMAL PROPERTIES OF SLUDGE AND COAL

In the study described here, granulated coal, with a maximum particle size of 9.5 mm, was added to liquid sludge in dosages ranging from 0.2 to 1.0 kg coal/kg dry sludge solids. This range was established from a theoretical basis of heat energy (Btu's) needed for combustion at different cake moisture contents. In Table 1, the measured thermal properties of the sludge and coal can be compared. In Table 2 are listed the heat energy required to evaporate the water and the auxiliary fuel required for several sludge cakes in a multiple hearth furnace. An exit gas temperature of 800°F was assumed for calculations. Note that the autogenous range is theoretically somewhere between 25 and 30 percent cake solids (assuming 64 percent volatile solids and 23,244 kJ/kg [10,000 Btu/lb] volatiles). Energy supplied by the sludge cake is dependent on the percent volatiles in the solids, and heat requirements are on the order of 5,927 kJ/kg (2,550 Btu/lb) of water at 75 percent excess air.

TABLE 1. COMPARISON OF THERMAL PROPERTIES

Property	Average values	
	Coal	Sludge
Btu/lb* dry volatile solids	14,200	10,500
Volatile solids, %	93.6	65-75
Btu/lb dry solids	13,300	8,100
Sulfur, %	0.60	0.60

*To convert to kJ/kg, multiply by 2.3244

The heat value of the coal is much higher (30,900 kJ/kg [13,300 Btu/lb]) than that of the sludge (18,800 kJ/kg [8,100 Btu/lb]). Therefore, adding coal will increase the volatile content and, also, the ratio of volatiles to water. As coal is added to the sludge (before filtration), the solids concentration is increased. This is passed on through vacuum filtration and results in a cake with less water per unit of volatiles. The experimental result of adding coal in varying dosages to the initial sludge is illustrated in Figure 1. The solids concentration of the cake produced on the vacuum filter is increased, but not linearly because of cake buildup and resistance to filtration. In Figure 2, coal dosage is plotted against the calculated heat requirement to incinerate sludge cake with the corresponding moisture content (see Figure 1). This heat must be provided by the combustion of the sludge solids in the dry cake and, if necessary, auxiliary fuel. Also shown in Figure 2 is the experimentally determined heat of combustion of coal-sludge mixtures (or energy supplied by the solids). It is evident from the diagram that when coal dosage is above 0.3 kg coal/kg sludge solids, there

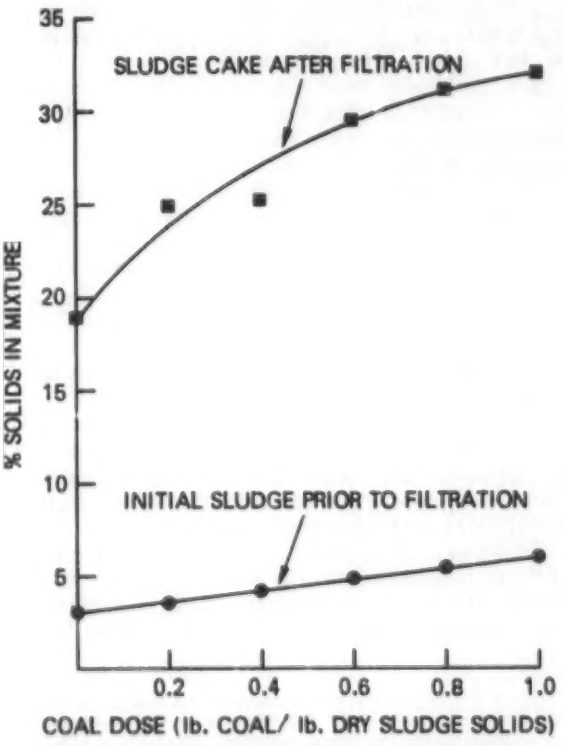

Figure 1. Effect of coal addition on solids content.

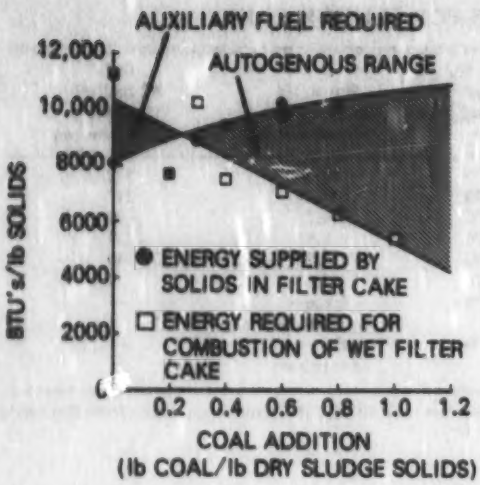

Figure 2. Incinerator requirements.

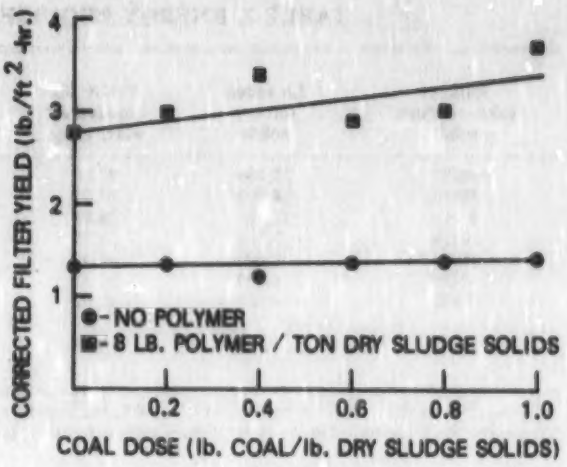

Figure 3. Effect of coal and polymer on filter yield.

will be sufficient energy from the combustion of the sludge cake to supply the heat needed for incineration. Further addition of coal will only produce excess heat, which is probably not usable.

COAL AS AN AID TO DEWATERING

Dewatering rates for typical chemically conditioned primary-activated sludge mixtures are in the order of 14.6 to 30 kg/m²-hr (3 to 6 lb/ft²-hr*).[5] Chemical costs range from about $5 to $16 metric ton[6] dry sludge solids. In Figure 3 are the results of filter leaf testing with mixed primary-activated sludge obtained from the USEPA pilot plant in Lebanon, Ohio. The cycle time simulating filter drum action was 6 minutes per revolution, with 30 percent of the total time for submergence and 70 percent of the time for drying under vacuum. The filter media was a polypropylene monofilament fabric. Results show that without polymer, coal addition has no effect at all on the net yield of sludge solids (i.e., mass of coal in the cake is subtracted out). With polymer added at the rate of 4.0 kg/metric ton, yield approximately doubled. Coal dosage now had a positive effect when polymer was added to the sludge. Net yield increased by a factor of 1.2 when coal dose was increased from 0 to 1.0 kg coal/kg sludge solids. Net yield at 1.0 kg coal/kg sludge was 17.5 kg/m²-hr (3.5 lb/ft²-hr). Total yield was 34 kg/m²-hr (7 lb/ft²-hr). At a cycle time of 6 minutes, this yield indicated a thick cake. Cycle time could easily be halved and a cake thick enough for easy discharge could be produced. Total and net yield would be expected to increase by a factor of $\sqrt{2}$ (i.e., 1.414) if cycle times were halved, but an undesirable reduction in cake solids would be anticipated.

*To convert from lb/ft²-hr to kg/m²-hr multiply by 4.882

It is clear from these data that the addition of coal in the particle size range used in this study does not produce a startling increase in filter yield. The primary advantage is the increase in cake solids noted earlier (Figure 1) and the reduction in auxiliary fuel requirements (Figure 2). It is possible that coal ground to a finer mesh size would have a greater influence on net filter yield.

COST IMPLICATIONS

Only general ranges can be discussed for costing the addition of coal to sludge. In terms of auxiliary fuel, the coal is effective at a dose of approximately 0.3 kg coal/kg solids. Current contract values for 1 metric ton of mine-run bituminous coal are in the range of $22-$33.[7] At $33/metric ton, the cost for coal addition to the sludge would be $10/metric ton dry sludge solids.

If no coal were used to condition the cake, the resultant moisture content of the product cake would be the 82 percent recorded for the no-coal sludge in Figure 2. From Table 2, we can see that about 220 liters (59 gallons) of No. 2 oil would be required to provide conditions of combustion for this cake. Using the low end of the current contract price range of 7¢-9¢/liter (27¢-33¢/gallon) results in a fuel cost of $17.50/metric ton of dry solids. Net savings in sludge processing cost would be $17.50 minus $10.00 or $7.50/metric ton.

CONCLUSIONS OF LAB TESTING

Addition of granulated coal to liquid sludge can result in a significant cost reduction. Although the coal does provide the necessary heat energy for autogenous combustion, it cannot, in itself, serve as a complete conditioning agent. Additional conditioning agents would still be required, but indications are

TABLE 2. ENERGY REQUIREMENTS FOR SLUDGE CAKE

Ratio of cake moisture/ solids	Lb water/ ton dry solids	Btu × 10_6 required for water evap.	Btu × 10_6* provided by sludge solids, ton	Btu × 10_6 auxiliary Btu requirement	Gal #2 fuel oil for auxiliary requirement/ton
95/5	38,000	97.00	14.70	82.30	572
90/10	18,000	45.95	14.70	31.25	217
85/15	11,333	28.93	14.70	14.23	99
82/18	9,111	23.25	14.70	8.55	59
80/20	8,000	20.42	14.70	5.72	40
75/25	6,000	15.31	14.70	0.61	4
70/30	4,667	11.91	14.70	Excess autogenous	—
65/35	3,714	9.49	14.70	Excess autogenous	—

*Assuming volatile solids of dry sludge solids = 70%; heating value of volatile solids = 10,500 Btu/lb volatiles; to convert from Btu's to kJ (kilojoules), multiply by 1.054; to convert from gallons to liters, multiply by 3.785.

that an interaction between the coal and the conditioning agent may result in a reduction in the need for conditioning chemicals when coal is used.

FUTURE STUDY

To completely define the use of coal as a sludge additive, actual tests with full scale vacuum filters are required. In the interim, pilot-scale testing with a small continuous rotary vacuum filter is being performed at the EPA Lebanon, Ohio, pilot plant. The operation of this machine will enable production of sludge cakes with varying parameters such as cycle time and granulated coal dosage.

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DO REGULATED FREIGHT RATES DISCOURAGE RECYCLING?

Oscar W. Albrecht*

BACKGROUND

The question of whether Interstate Commerce Commission (ICC) regulated freight rates for common carriers are discriminatory and discourage the movement of waste materials for recycling has been debated publicly at length, particularly in regard to the railroad movement of ferrous scrap. The Institute of Scrap Iron and Steel (ISIS) has repeatedly charged that railroad rates favor the movement and use of the virgin material (iron ore) over ferrous scrap in the making of steel, asserting that railroad freight rates average 2½ times more for iron and steel scrap than for iron ore.[1] The National Association of Recycling Industries (NARI) has also argued that waste materials destined for recycling have been discriminated against by the regulated rates.[2]

The issue of fair and appropriate freight rates is not a new one. In 1925, for example, when agriculture was considered in a state of economic depression, the "Hoch-Smith Resolution" passed by Congress stated ". . . it is declared to be the true policy in rate making to be pursued by the Interstate Commerce Commission in adjusting freight rates, that the conditions which at any given time prevail in our several industries should be considered insofar as it is legally possible to do so, to the end that commodities may freely move."[3] There is precedent, therefore, for setting freight rates to achieve national goals other than maintaining a healthy and economically viable transportation system.

The increasing interest in resource recovery and recycling has spurred new interest in the ICC's rate making procedures. A study conducted for the Department of Transportation examined rates and other transportation characteristics involved in the movement of secondary materials vis-a-vis basic raw materials.[4] The results were inconclusive, but the study pointed out that ICC's accounting and costing procedures were generally inadequate for determining the cost of railroad transport. The ICC recently initiated its own investigation of the railroad freight rate structure and is studying what base to use in determining rate of return.

RESULTS OF TWO EPA STUDIES

In 1965, Congress enacted the Solid Waste Disposal Act to deal with the problem of accumulating solid waste. The Act was subsequently amended by passage of the Resource Recovery Act of 1970. The latter required that study and investigation be made of the effect of existing public policies upon the reuse, recycling, and conservation of materials from solid waste.[5]

The purpose of this paper is to review the results of two U.S. Environmental Protection Agency (EPA) funded efforts directed at investigating alleged differences in regulated rates for virgin and secondary materials. The studies were carried out by the Research Planning Institute (RPI) and Moshman Associates.[6,7] The RPI study was concerned mainly with the manner in which discriminatory rates influence long-run investment decisions in the steel and papermaking industries. It assumed that technological and economic constraints limit the choice of raw materials in the short-run, but that raw material costs and transportation rates play an important role in long-run decisions as to where and what kinds of plants to build.

The RPI findings showed that on a direct comparison of the revenues generated, the rail movement of virgin materials was favored over that of secondary materials. If transport costs were weighted by chemically equivalent amounts needed to produce the

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final products (steel and paperboard), however, the secondary materials (ferrous scrap and waste paper) had the advantage if movement was effected by rail, and the virgin materials (woodchips and residue) benefitted if movement was made by motor carrier (Table 1).

TABLE 1. COMPARISON OF REVENUES FROM TRANSPORTING VIRGIN AND SECONDARY MATERIALS, 1970*

Mode	Virgin material ($/net ton)	Secondary material ($/net ton)
Rail transport — direct comparison		
iron ore vs. ferrous scrap	2.31	4.83
roundwood vs. waste paper	1.92	8.01
woodchips and residue vs. waste paper	2.58	8.01
Rail transport — chemically weighted†		
coal basis vs. ferrous scrap‡	7.34	4.83
coke basis vs. ferrous scrap ‖	7.79	4.83
woodchips and residue vs. waste paper	9.47	8.81
pulp vs. waste paper	16.81	8.81
Motor transport — direct comparison		
woodchips and residue vs. waste paper	2.44	9.68
pulp vs. waste paper	5.10	9.68
Motor transport — chemically weighted†		
woodchips and residue vs. waste paper	8.95	10.65
pulp vs. waste paper	5.10	10.65

*Source: Reference No. 6

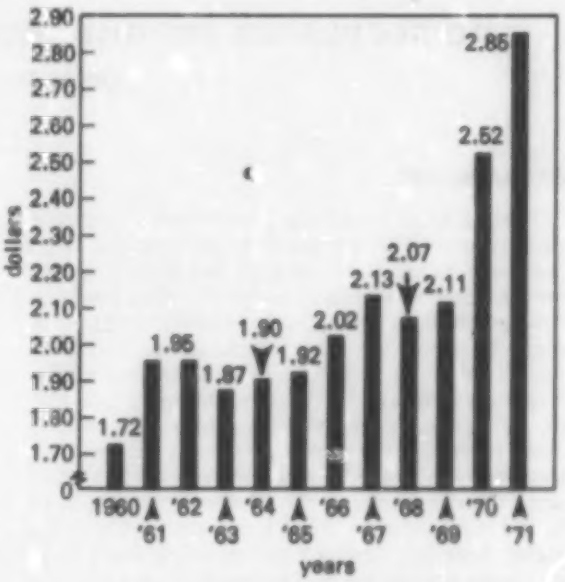
†Chemically weighted by the transportation costs for component amounts needed to produce 1 ton of raw steel or 1 ton of paperboard.

‡The coal basis assumes that coal is moved to the blast furnace for conversion into coke; transport of limestone is included.

‖The coke basis assumes that coke is moved to the blast furnace from distant coking ovens; transport of limestone is included.

A basic weakness of the RIP study is that it uses average revenue per ton (conversely, cost to shipper) rather than actual ton-mile rates. The first value is obtained by dividing the carrier's revenue for transporting each commodity by the total tonnage, a technique that is insufficient to show rate discrimination. It would be better to use actual ton-mile rates for individual commodity shipments, but such data are difficult to obtain because of the complexity of freight rate schedules. Average revenue per ton is more inclusive of shippers costs, but does not relate to any particular service because it lumps together various sizes of shipments and dissimilar distances (routes). The comparisons are thus valid only if carrier costs are similar for each type of service.

The Moshman study compared the contribution in freight rates of secondary material with that of virgin material. Contribution was defined as the difference between the rate for the shipment and the associated variable costs. Comparisons were made for six commodity pairs: ferrous scrap and iron ore, cullet and glass sand, aluminum scrap and aluminum ingot, waste paper and wood pulp, scrap rubber and new rubber, and reclaimed rubber and new rubber. The matching process for similarities in length of haul and carload weights greatly reduced the number of possible comparisons, and it was necessary to compare "all moves" for most commodities.



Additional railroad revenue from transporting ferrous scrap compared with that from iron ore (¢/ton). (Source: Reference No. 6.)

Results of the Moshman study indicated that ferrous scrap, glass cullet, and reclaimed rubber were making larger contributions than their virgin counterparts (Table 2). The national averages, however, obscured many individual variations, including those between the railroad territories and between different lengths of routes. For example, ferrous scrap made a larger contribution than iron ore on a national basis, but most scrap moved less than 200 miles; within this distance, the contribution rates of iron ore and scrap were about equal.

WHEN ARE RATES DISCRIMINATORY?

A basic question relating to the results of the above studies is whether the differences in revenues or differences in contributions to variable costs can be accepted as evidence of discriminatory rates. In economic terminology, discrimination exists when a seller sells an identical product to different buyers at different prices. When variations in production or

TABLE 2. CONTRIBUTIONS TO VARIABLE COSTS RESULTING FROM RAILROAD RATES FOR SELECTED VIRGIN AND SECONDARY MATERIALS, 1969*

Commodity	Contribution to variable cost (%)
Iron ore	171
Ferrous scrap	196
Wood pulp	187
Waste paper	144
Glass sand	130
Glass cullet	204
Aluminum ingot	55
Aluminum scrap	38
New rubber (natural and synthetic)	52
Scrap rubber	40
Reclaimed rubber	66

*Source: Reference No. 7

service costs are taken into account, discrimination exists when a seller charges prices that result in different price-to-incremental-cost ratios. If P_1 and MC_1 are the price and incremental cost for one product and P_2 and MC_2 are the price and incremental cost of a second product, economic discrimination exists when P_1/MC_1 is not equal to P_2/MC_2.

This rule for evaluating discrimination is appropriate where conditions of competitive equilibrium exist and product prices tend to equal incremental cost of production. For regulated industries or utilities operating under conditions of decreasing costs, the rule is less applicable. It is generally recognized that railroads have unused track capacity and that additional handling capacity can be added at less cost per ton than the initial cost of the basic track. This means that for a considerably larger volume of traffic the average ton-mile cost would decline and the incremental cost would be below the average total cost. Thus, conventional cost pricing, where price is equal to incremental cost, is inappropriate for setting rates if the railroads are to operate without losses.

Given the nature of the cost functions and the tradition of government regulation, the question is how can railroad rates be efficiently structured to serve the public interest? The Emergency Railroad Transportation Act of 1933, which the ICC uses as justification for economic discrimination in rates, states: "In the exercise of its power to prescribe just and reasonable rates the Commission shall give due consideration, among other factors, to the effect of rates on the movement of traffic by the carrier or carriers for which the rates are prescribed; to the need, in the public interest, of adequate and efficient railway transportation service at the lowest cost consistent with the furnishing of such service; and to the need of revenues sufficient to enable the carriers, under honest, economical, and efficient management to provide such service."[8] Thus, discrimination in the ICC sense refers to a rate that adversely affects the movement of a commodity or its contribution relative to another rate. Since regulations require railroads to operate some unprofitable lines, the ICC allows economic discrimination to enable carriers to recapture their total costs.

The issue of discrimination is inextricable from the question of competition. Section 3 of the ICC Act of 1887, while not specifically referring to discrimination, makes it unlawful for a railroad to give any "... undue or unreasonable preference or advantage to any particular person, company, firm, ..." or to subject any of the preceding to ". . . undue or unreasonable prejudice or disadvantage. ..."[9] It is said that the ICC has vacillated with regard to the question of competition between iron ore and ferrous scrap and has remained noncommittal with regard to the other commodities. If competition exists, the Commission may still approve rate differentials for one or more of the following reasons: (a) cost of service differs, (b) value of service differs, or (c) intermodal competition is present. Thus, it is impossible to conclude that discrimination exists based on rate disparity alone.

SUMMARY

The two EPA studies failed to provide conclusive evidence that ICC-regulated rates discriminated against secondary materials. The RPI study implied that discrimination might exist (in the ICC sense) for motor carrier rates weighted by the amounts of chemical constituents needed to make paperboard. The study failed to document, however, that long-run investment decisions involving raw material substitutions and plant locations were actually affected by the rate differences. The Moshman study compared percentage contributions to variable cost of different rates per ton-mile. To the extent that the costs were representative, the study implied that economic discrimination existed for ferrous scrap because the applicable rates did vary from incremental costs on long hauls when compared with those for iron ore. Discrimination was also suggested for cullet and reclaimed rubber because the contribution percentages for these materials were higher than for their virgin counterparts. However, as mentioned earlier, the appropriateness of ICC costing procedures has been questioned.

Further research is needed to assess fully the effects of differential rates for virgin and secondary materials. It would be particularly desirable to

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examine the impact of different rates on the movement of secondary materials from more distant areas that may be disadvantaged by higher transport costs relative to total production costs. The role of transport costs in management decisions concerning the choice of raw materials and location of plants should also be given further study.

Concerning the basic question of rate discrimination, the issue is related to the national goals and objectives for regulating the common carriers. Probably in no country of the world today are transportation rates entirely free of government regulation, and in many countries, the rates reflect social goals unrelated to the cost of supplying transport service. The point to be emphasized is that at present there are no clearly defined criteria for establishing freight rates. And determination of discrimination in the conventional economic sense, based on incremental cost pricing, is generally inappropriate because of the nature of the costs prevailing in the railroad industry. If freight rates are to be regulated to achieve a social objective, such as recycling, new decision rules to guide the rate making processes are needed.

Until recently, the ICC has rather consistently resisted pressures to make rate adjustments in favor of social goals, but there are indications that it may now be relenting somewhat in the face of increased public pressure. Since the rates in the EPA studies were analyzed, there have been at least four general rate increases, and for several of these the ICC ordered "holddowns" on selected recyclable commodities.

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